Species Status Assessment Report
for the
MacGillivray's Seaside Sparrow
(*Ammodramus maritimus macgillivraii*)

Version 1.3

Credit: A. Schwarzer

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EXECUTIVE SUMMARY

This report summarizes the results of a Species Status Assessment completed for the MacGillivray's seaside sparrow (*Ammodramus maritimus macgillivraii*) to assess the subspecies' overall viability. The MacGillivray's seaside sparrow is a subspecies of seaside sparrow that occurs in Atlantic coastal marshes in South Carolina, Georgia, and Florida. In this assessment, we defined a population of MacGillivray's seaside sparrow as a group of inter-breeding individuals that live in the same place at the same time. Birds in South Carolina are widespread, but exhibit a scattered and patchy distribution where birds are locally abundant at just a few sites, or occur in substantially lower densities or not at all outside of the high abundance areas. In Georgia and northeast Florida, MacGillivray's seaside sparrows also are widespread, but are distributed more uniformly along the states' coastlines. Based on the subspecies' distribution pattern from survey data and the connectivity of marsh habitat, we identified four breeding populations. Three populations occur in South Carolina: North Inlet-Bulls Island; Headquarters Island; and ACE Basin. The fourth population, Georgia-Florida, is a relatively contiguous single population. MacGillivray's seaside sparrow once occupied approximately 370 miles (595 kilometers) of coastline in the southeastern U.S. and is believed to be extirpated from approximately 27% (100 miles or 160 kilometers) of that historical range. Areas from which the subspecies has been extirpated are all in Florida, including Duval County south of the City of Jacksonville, and St. Johns, Flagler, and Volusia Counties.

To evaluate the viability of the MacGillivray's seaside sparrow, we characterized the needs, estimated the current condition, and predicted the future condition of the subspecies' in terms of resiliency, representation, and redundancy. Briefly, resiliency describes the ability of a population to withstand environmental and demographic stochasticity; representation describes the ability of the subspecies to adapt over time to long-term changes in the environment (for example, climate change); and redundancy describes the ability of the subspecies to withstand catastrophic events (for example, droughts, hurricanes). In general, the more redundant and resilient a subspecies is and the more representation it has, the more likely the subspecies will sustain populations over time, even under changing environmental conditions.

MacGillivray's seaside sparrows spend their entire life in coastal salt and brackish marshes in South Carolina, Georgia, and Florida in the United States. In South Carolina, individuals breed in lower elevation areas of high marsh and in managed impoundments which are often brackish and non-tidal. Birds in Georgia and Florida breed in higher elevation areas of natural low marsh that experience semidiurnal tides. MacGillivray's seaside sparrow is generally considered a non-migratory, sedentary bird, but anecdotal data and historical accounts suggest individuals exhibit local seasonal movements (between breeding and wintering seasons).

Primary limiting factors for seaside sparrow survival and reproduction are predation, flooding, and extreme weather events. For eggs and nestlings, MacGillivray's seaside sparrows need a nest cup that often has a canopy made from interweaving surrounding marsh grasses to provide shelter from predation. Adult MacGillivray's seaside sparrows have behavioral adaptations to balance the trade-off in risk from predation and flooding, and therefore, will shift nest-site placement along a nest height gradient to contend with these dual-risks. For example, by placing

a nest lower to the surface in dense vegetation, the amount of cover to hide the nest from predators will increase; however, this will also increase the probability of the nest flooding. In addition, MacGillivray's seaside sparrows avoid marshes that are near uplands, which may be an additional strategy to minimize predation risk since forests serve as sources of predators.

Our analysis of the past, current, and future influences on the MacGillivray's seaside sparrow needs for long term viability revealed that there are two primary factors, predation and tidal flooding, that pose the largest risk to future viability of the subspecies. Predation is a significant mortality factor for MacGillivray's seaside sparrow nests. In recent demographic studies of MacGillivray's seaside sparrow, predation was the primary source of nest mortality with 28%, 63%, and 64% of known-fate nests lost to predators in South Carolina, Georgia, and Florida, respectively. Predators of seaside sparrow nests include raccoons, American mink, rice rats, fish crows, red-winged blackbirds, boat-tailed grackles, marsh wrens, northern harriers, and snakes. The most common nest predators that nest sympatrically in marshes with seaside sparrows are rice rats, red-winged blackbirds, and marsh wrens. While little is known about predation rates on juveniles, predation on adults during the breeding season appears to be uncommon. The primary predators of juvenile and adult seaside sparrows are avian species, including northern harrier, American crow, and owls.

MacGillivray's seaside sparrows are affected by tidal flooding during all stages of their life history, although the greatest demographic impact is on nests. In Georgia, 17% of nests in one marsh were lost to flooding from 2013-2015, and 6% of nests were lost to flooding at a marsh in Florida from 2015-2017. Despite the low number of direct nest mortalities from flooding, larger tidal amplitudes of high tides had a strong negative effect on daily nest survival probability in Florida, indicating that there may be secondary impacts from tidal flooding such as increased concentrations of predators in breeding areas during the highest high tides. While tidal flooding is an important cause of nest failure for seaside sparrows, in addition to predation, the relative importance of tidal flooding versus predation is highly variable from year to year. As stated previously, MacGillivray's seaside sparrows face a trade-off in nest site selection based on the competing risks of nest predation and nest flooding. Placing nests higher from the ground to avoid flooding may increase the risk of nest predation due to lack of cover. Sea-level rise is predicted to change the tidal regime by increasing the frequency and severity of tidal flooding, and MacGillivray's seaside sparrows may have very low rates of nest success under extreme sea-level rise, especially when combined with high nest predation.

As discussed above, predation and tidal flooding are the primary factors currently influencing the viability of MacGillivray's seaside sparrow. We evaluated the current condition of MacGillivray's seaside sparrow by assessing demographic and habitat metrics of resiliency, representation, and redundancy in relation to these factors. Given data availability, MacGillivray's seaside sparrow resiliency was estimated using population-level demographics (abundance, reproduction, and population growth). Abundance and reproductive values were estimated from existing survey and demographic data from across the subspecies' range. To assess current population resiliency, we estimated the maximum number of birds detected at a survey point (as a proxy for abundance), nest predation rates, nest inundation rates, daily/period nest survival rates, and population growth rates by analysis unit. To further assess potential population growth rates for the South Carolina, Georgia, and Florida analysis units, the Florida

Fish and Wildlife Conservation Commission (FWC) conducted an additional analysis to simulate population growth rates (λ) for each analysis unit.

Our results indicated that MacGillivray's seaside sparrow populations in South Carolina are likely stable and have higher resiliency than the Georgia-Florida population. Abundance was higher in the Georgia-Florida population (2.5-3.3 mean number of birds per occupied point) compared to the South Carolina populations (1.0-2.0 birds per occupied point), but nest survival was lower in Georgia and Florida than in South Carolina. Nest survival is a known reproductive constraint for passerines and low nest survival is a leading driver of population declines. When considering the ranges of the estimated population growth rates, the Georgia-Florida population is experiencing negative population growth, leading to an overall decline in the current resiliency of this MacGillivray's seaside sparrow population and below a threshold which is needed to maintain population stability (i.e., λ = 1.0). This is further supported by the additional FWC modeling which found that the Georgia and Florida analysis units are currently undergoing population declines. On the other hand, based on higher nest survival rates and stable population growth rates, the North Inlet-Bulls Island population in South Carolina has higher resiliency than the Georgia-Florida population. Current nesting data were unavailable for the two other populations in South Carolina (i.e., Headquarters Island and ACE Basin) so we were unable to estimate population growth rates; however, survey data and anecdotal observations indicate that MacGillivray's seaside sparrows in these populations are present at similar abundances and nest in dense concentrations as found in the North Inlet-Bulls Island population. Therefore, we infer that the Headquarters Island and ACE Basin populations also have higher resiliency than the Georgia-Florida population and are likely stable in their current state.

To assess the current representation of the subspecies, we used three metrics to estimate and predict representative units that reflect the subspecies' adaptive capacity: 1) genetic variability, 2) habitat variability, and 3) tidal variability. The subspecies should have resilient populations across the representative units to maintain existing adaptive capacity. There are currently three genetically distinct representative units of MacGillivray's seaside sparrow across its range: 1) the Headquarters Island population, 2) the North Inlet-Bulls Island population, and 3) the Georgia-Florida population. Additional representative units include three habitat types: two habitat types used by MacGillivray's seaside sparrow in South Carolina and one habitat type used by the subspecies in Georgia-Florida. The Georgia-Florida population breeds in a different marsh habitat type, higher elevation low salt marshes, than the three populations that occur in South Carolina. For the Headquarters Island population in South Carolina, birds occupy only low elevation high marsh habitats. MacGillivray's seaside sparrows in the ACE Basin and North Inlet-Bulls Island populations occur in both low elevation high marsh and managed impoundments. The final representative unit for MacGillivray's seaside sparrow is tidal variability, which reflects the subspecies' ability to utilize habitat that experiences a range of tidal fluctuation. Birds that inhabit natural coastal marshes encounter tides ranging from approximately four to ten feet in elevation, while birds that inhabit managed impoundments do not experience tidal fluctuations.

For redundancy, we evaluated the current distribution of MacGillivray's seaside sparrow populations through their present-day geographic locations. To have a high level of redundancy, MacGillivray's seaside sparrow would need to have resilient populations spread throughout its

range. The subspecies currently occurs in approximately 73% of its known range in South Carolina, Georgia, and northeast Florida. There are currently four populations spread throughout this range, except the extirpated zone in Florida. However, the subspecies' redundancy is reduced due to (1) all three populations with greater resiliency (the South Carolina populations) being limited to the northernmost third of the currently occupied range and (2) the downward trend in resiliency for the largest population, Georgia-Florida.

In the future, predation and tidal flooding remain the two primary factors influencing the viability of MacGillivray's seaside sparrow. Climate change, specifically inundation due to sea level rise, and habitat degradation/conversion due to urbanization are other factors that were considered to potentially influence the viability of the subspecies. To predict future resiliency of MacGillivray's seaside sparrow, we used an individual-based model to forecast changes in nest success (demographic factor) for the Georgia-Florida population to understand the risk from predation and inundation due to sea level rise and increased storm events. This model was used to forecast nest success, nest predation, and nest inundation from an observed, historical flood-risk trend (1979-2015) to the year 2113 for the Georgia analysis unit of the Georgia-Florida population of MacGillivray's seaside sparrow given a range of plausible, future scenarios. Because of the lack of historical and current population estimates for MacGillivray's seaside sparrow, changes in breeding habitat area were used as a surrogate metric to evaluate the future condition of subspecies. Therefore, we used a model to forecast high abundance breeding habitat (habitat factor) to understand the risk from inundation due to sea level rise and urbanization. We predicted the change in high abundance breeding habitat for the Georgia and South Carolina analysis units from its current condition to years 2025, 2050, 2075, and 2100. We also predicted the future population size of the Georgia analysis unit by projecting changes in the quantity and spatial extent of high abundance breeding habitat to years 2025, 2050, 2075, and 2100.

Results from the individual-based model for MacGillivray's seaside sparrow indicated that predation-risk had a greater influence on daily nest survival rate than flood-risk, even when flood-risk was simulated to increase with sea-level rise. MacGillivray's seaside sparrow nest-height selection behaviors had an overall positive effect on nest survival, and therefore, in terms of nest success, these behaviors will likely contribute at some level to the Georgia-Florida population's resiliency to predation and sea-level rise. Results from the modelling that incorporated increased storm frequency indicated that impacts to nesting success from increased storm frequency were outweighed by the impacts from increased flood-risk due to sea-level rise and present-day levels of predation. In future scenarios of sea-level rise, daily nest survival is reduced to less than 80% under high predation-risk. Low nest survival is a leading driver of population declines, and under the current condition, we determined that low nest survival is a major factor in the downward trend for MacGillivray's seaside sparrow in the Georgia-Florida population. Therefore, in the future, a continuing reduction in daily nest survival will lead to lower period nest survival resulting in even larger negative population growth rates (and accompanying declines in resiliency) for the Georgia-Florida population of the subspecies.

Results from the change in high abundance breeding habitat models for MacGillivray's seaside sparrow projected a loss in the majority of high abundance breeding habitat across the subspecies' range by 2100, from inundation due to sea-level rise. The large anticipated declines in habitat availability will reduce the ability of three MacGillivray's seaside sparrow populations

(North Inlet-Bulls Island, Headquarters Island, and Georgia-Florida) to withstand stochastic events, thereby reducing resiliency. In South Carolina, the North Inlet-Bulls Island population is projected to lose 30% to 44% of its high abundance breeding habitat by 2050, and 77% to 88% by 2100, while the ACE Basin population is projected to gain 7% by 2050, but lose 11% by 2100. Based on the location of the Headquarters Island population and the tidally-influenced natural marsh that the subspecies uses there, we infer that the Headquarters Island population also will experience a loss in high abundance breeding habitat from sea level rise. For the North Inlet-Bulls Island and Headquarters Island populations, future projected losses of high abundance breeding habitat will likely result in population changes from stable to declining population growth, thereby reducing future resiliency. The Georgia-Florida population may lose on the order of 80% of its high abundance breeding habitat by 2100 (27% by 2050). In addition, the MacGillivray's seaside sparrow population size for the Georgia analysis unit was forecast to decline by 54% by the year 2100, from 32,800 birds to 15,088 birds, based on the loss of high abundance breeding habitat (Hunter et al. 2016a, p. 6). Estimated future reductions in MacGillivray's seaside sparrow nest survival, compounded with future projections of habitat loss, will exacerbate the current declining trend in resiliency for the Georgia-Florida population.

With the loss of high abundance breeding habitat across the range of the subspecies, the three genetically distinct units will likely remain into the future but with reduced resiliency for all three representative units (Headquarters Island population, North Inlet-Bulls Island population, and Georgia-Florida population). The South Carolina populations will remain isolated with limited to no connectivity as already represented by their current distribution across the state. The Georgia-Florida population may become isolated over time with habitat patches becoming increasingly fragmented under sea-level rise and habitat quality diminishing from increasing isolation. Despite the use of variable habitats by MacGillivray's seaside sparrow, there are projected declines of high abundance breeding habitat throughout the subspecies' range and for all three habitat variability representative units. In addition, tidal regimes are projected to change under sea-level rise and from increased storm frequency and severity, which will alter nesting conditions for MacGillivray's seaside sparrow. While the subspecies exhibits nest-site selection behaviors in response to perceived flooding and predation risks, changes in regular tidal cycles may occur on a temporal scale that is faster than MacGillivray's seaside sparrows can adapt to. Due to the increased risk of genetic isolation into the future from high abundance habitat loss, which affects all three different habitat types used by MacGillivray's seaside sparrow, and due to the diminished ability of the subspecies to nest successfully because of changes related to sea-level rise, we expect future representation to decline.

In the future, redundancy will be reduced because of the loss of high abundance breeding habitat across the range resulting in three of the four MacGillivray's seaside sparrow populations having reduced resiliency (North Inlet-Bulls Island, Headquarters Island, and Georgia-Florida). For the Georgia-Florida population, the current declining trend in resiliency will worsen due to reductions in habitat and nest survival. In addition, habitat loss will result in increased habitat fragmentation (further reducing redundancy) leading to isolated pockets of birds along the major river estuaries (particularly Savannah, Altamaha, and Satilla) instead of a single contiguous population as it is currently distributed. Any existing connectivity between MacGillivray's seaside sparrow populations will likely remain low or become further reduced as habitat patches become increasingly fragmented with sea-level rise.

TABLE OF CONTENTS

LIST OF TABLES

LIST OF FIGURES

CHAPTER 1 – INTRODUCTION

The MacGillivray's seaside sparrow (*Ammodramus maritimus macgillivraii*) is a subspecies of seaside sparrow that occurs in Atlantic coastal marshes in South Carolina, Georgia, and Florida. We, the U.S. Fish and Wildlife Service (Service), were petitioned to list the MacGillivray's seaside sparrow as endangered or threatened under the Endangered Species Act of 1973, as amended (16 U.S.C. 1531-1543) (Act), in April 2010 as a part of the Petition to List 404 Aquatic, Riparian and Wetland Species from the Southeastern United States by the Center for Biological Diversity (Center for Biological Diversity, 2010, p. 106). In September 2011, the Service published a 90-day finding that the petition presented substantial scientific or commercial information indicating that listing may be warranted for 374 species, including MacGillivray's seaside sparrow (76 FR 59836, September 27, 2011). A subsequent notice of violation for not meeting the statutory petition 12-month finding deadline was filed by the Center for Biological Diversity on February 20, 2013. The Service committed to a deadline of September 30, 2018 for submitting to the Federal Register a 12-month finding on MacGillivray's seaside sparrow. Therefore, a review of the status of the subspecies was initiated to determine if the petitioned action is warranted. Based on the status review, the Service will issue a 12-month finding for the MacGillivray's seaside sparrow. Thus, we conducted a Species Status Assessment (SSA) to compile the best available data regarding the subspecies' biology and factors that influence the subspecies' viability. The MacGillivray's seaside sparrow SSA Report is a summary of the information assembled and reviewed by the Service and incorporates the best scientific and commercial data available. This SSA Report documents the results of the comprehensive status review for the MacGillivray's seaside sparrow and serves as the biological underpinning of the Service's forthcoming decision (12-month finding) on whether the subspecies warrants protection under the Act.

The SSA framework (USFWS, 2016a, entire) is intended to be an in-depth review of the subspecies' biology and threats, an evaluation of its biological status, and an assessment of the resources and conditions needed to maintain long-term viability. The intent is for the SSA Report to be easily updated as new information becomes available and to support all functions of the Ecological Services Program of the Service, from Candidate Assessment to Listing to Consultations to Recovery. As such, the SSA Report will be a living document that may be used to inform Endangered Species Act decision making, such as listing, recovery, Section 7, Section 10, and reclassification decisions (the former four decision types are only relevant should the subspecies warrant listing under the Act). Therefore, we have developed this SSA Report to summarize the most relevant information regarding life history, biology, and considerations of current and future risk factors facing the MacGillivray's seaside sparrow. In addition, we forecast the possible response of the subspecies to various future risk factors and environmental conditions to formulate a complete risk profile for the MacGillivray's seaside sparrow.

The objective of this SSA is to thoroughly describe the viability of the MacGillivray's seaside sparrow based on the best scientific and commercial information available. Through this description, we determined what the subspecies needs to support viable populations, its current condition in terms of those needs, and its forecasted future condition under plausible future scenarios. In conducting this analysis, we took into consideration the likely changes that are

happening in the environment – past, current, and future – to help us understand what factors drive the viability of the subspecies.

For the purpose of this assessment, we define **viability** as a description of the ability of a subspecies to sustain populations in the wild beyond a biologically meaningful time frame. Viability is not a specific state, but rather a continuous measure of the likelihood that the subspecies will sustain populations over time (USFWS, 2016a, p. 9). Using the SSA framework (Figure 1-1), we consider what the subspecies needs to maintain viability by characterizing the status of the subspecies in terms of its **resiliency**, **representation**, and **redundancy** (USFWS, 2016a, entire).

Figure 1-1. Species Status Assessment Framework

- **Resiliency** describes the ability of a population to withstand stochastic disturbance. Stochastic events are those arising from random factors such as weather, flooding, or fire. Resiliency is positively related to population size and growth rate and may be influenced by connectivity among populations. Generally speaking, populations need enough individuals, within habitat patches of adequate area and quality, to maintain survival and reproduction in spite of disturbance. Resiliency is measured using metrics that describe population condition and habitat; in the case of the MacGillivray's seaside sparrow, we used abundance, nest survival rates, habitat quality, and habitat availability to assess resiliency.

- **Representation** describes the ability of the subspecies to adapt to changing environmental conditions over time. Representation can be measured through the genetic diversity within and among populations and the ecological diversity (also called environmental variation or diversity) of populations across the subspecies' range. Theoretically, the more representation the subspecies has, the higher its potential of adapting to changes (natural or human caused) in its environment. Genetic, habitat, and tidal variability were used to assess representation for the MacGillivray's seaside sparrow.

- **Redundancy** describes the ability of a subspecies to withstand catastrophic events. A catastrophic event is defined here as a rare, destructive event or episode involving multiple populations and occurring suddenly. Redundancy is about spreading risk among populations, and thus, is assessed by characterizing the number of resilient populations across a species' (or subspecies') range. The more resilient populations the subspecies has, distributed over a larger area, the better the chances that the subspecies can withstand catastrophic events. For the MacGillivray's seaside sparrow, we used the number of resilient populations, and the geographic distribution of those populations, to measure redundancy.
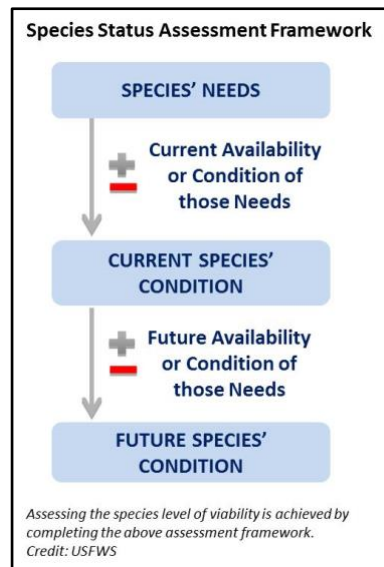
To evaluate the viability of the MacGillivray's seaside sparrow, we estimated and predicted the current and future condition of the subspecies' in terms of resiliency, representation, and redundancy.

This SSA Report includes the following chapters:

1. Introduction;
2. Subspecies Biology and Individual Needs. The life history of the subspecies and resource needs of individuals;
3. Factors Influencing Viability. A description of likely causal mechanisms, and their relative degree of impact, on the status of the subspecies;
4. Population and Subspecies Needs and Current Condition. A description of what the subspecies needs across its range for viability, and estimates of the subspecies' current range and condition; and,
5. Future Conditions and Viability. Descriptions of plausible future scenarios, and predictions of their influence, on MacGillivray's seaside sparrow resiliency, representation, and redundancy.

Cited literature can be found after the final chapter. Preliminary results from a genetic assessment of the subspecies, that are relevant to the methods and results described herein, are described, in detail, in Appendix A. An analysis of the distribution and abundance of the subspecies in South Carolina is presented in Appendix B. Results from population growth rate simulations across the subspecies' range are found in Appendix C.

This SSA Report provides a thorough assessment of the biology and natural history and assesses demographic risks, stressors, and limiting factors in the context of determining the viability and risks of extinction for the MacGillivray's seaside sparrow. Importantly, this SSA Report does not result in, nor predetermine, any decisions by the Service under the Act. In the case of the MacGillivray's seaside sparrow, the SSA Report does not determine whether the MacGillivray's seaside sparrow warrants protections of the Act, or whether it should be proposed for listing as a threatened or endangered species under the Act. That decision will be made by the Service after reviewing this document, along with the supporting analysis, any other relevant scientific information, and all applicable laws, regulations, and policies. The results of the decision will be announced in the *Federal Register*. The contents of this SSA Report provide an objective, scientific review of the available information related to the biological status of the MacGillivray's seaside sparrow.

CHAPTER 2 – SUBSPECIES BIOLOGY AND INDIVIDUAL NEEDS

In this chapter, we provide biological information about the MacGillivray's seaside sparrow, including its taxonomic history, morphological description, historical and current distribution and range, and known life history. We then outline the resource needs of individuals.

2.1 Taxonomy

The seaside sparrow (*Ammodramus maritimus*), which ranges from New England to southern Texas, was first described by Alexander Wilson in 1811 (pp. 68-69) as the sea-side finch, *Fringilla maritima,* from specimens collected along the coast of southern New Jersey. The species has had a complex taxonomic history, complicated by the difficulty of interpreting subtle plumage characteristics, the variability of within-population plumage variation, and the limited number of specimens in fresh basic plumage in museum collections (Woltmann, et al., 2014, pp. 1-2). Currently, there are seven extant subspecies of seaside sparrows generally recognized (Post & Greenlaw, 2009, unpaginated). These seven subspecies are divided into two phylogenetically distinct groups, an Atlantic Coast group and a Gulf Coast group (Avise & Nelson, 1989, p. 648). The subspecies, MacGillivray's seaside sparrow (*Ammodramus maritimus macgillivraii*), falls into the Atlantic Coast group and has been described as occurring from North Carolina to northern Florida. First described by John James Audubon in 1834, as a full species, *Fringilla macgillivraii,* from the coastal salt marshes of Charleston, South Carolina (p. 285), it has been considered a subspecies of seaside sparrow since at least 1899 (American Ornithologists' Union [AOU], 1899, pp. 118-119).

Other subspecies within the Atlantic Coast group include the northern seaside sparrow (*A. m. maritimus*), which breeds from New Hampshire south to northern North Carolina and winters from Virginia south to northeast Florida; the extinct dusky seaside sparrow (*A. m. nigrescens*) which was a resident of coastal mid-Atlantic Florida; and the endangered Cape Sable seaside sparrow (*A. m. mirabilis*) a resident of the Everglades region of Florida (Post & Greenlaw, 2009, unpaginated). There were several other subspecies described in the early twentieth century, including *A. m. waynei* (originally described *Thryospiza maritima waynei*; Oberholser, 1931, pp. 125-126), *A. m. shannoni* (*T. maritima shannoni*; Bailey, 1931, p. 1 in McDonald, 1988, p. 11), and *A. m. pelonotus* (*T. maritima pelonota*; Oberholser, 1931, pp. 126-127). Both *A. m. waynei* and *A. m. shannoni* were synonymized with *A. m. macgillivraii*; however, there has been considerably more confusion about *A. m. pelonotus*. *A. m. pelonotus* was first described as the Smyrna seaside sparrow from New Smyrna Beach, Florida (Oberholser, 1931, pp. 126-127). The subspecies was thought to have ranged from south of Jacksonville, Florida north to the Florida-Georgia state line (Kale, 1983, pp. 42-44). After much debate and an extensive taxonomic review of the species' assemblage, it was proposed that three of the Atlantic Coast subspecies be combined (i.e., *A. m. maritimus*, *A. m. macgillivraii*, and *A. m. pelonotus*; McDonald, 1988, p. 20). The recommendation to combine all three of these subspecies has not been accepted in full; however, seaside sparrow experts do consider *A. m. pelonotus* to be synonymous with *A. m. macgillivraii* (Kale, 1996, p. 608; Rising, 2005, p. 493).

The reclassification of *A. m. pelonotus* with *A. m. macgillivraii* has not been officially assessed by the American Ornithological Society (AOS; formerly the AOU), which publishes the Checklist of North American Birds. The last published list to include subspecies was published in 1957 and more recent editions do not include the treatment of subspecies (AOU, 1957, entire; AOU, 1998, entire). Because a complete revision of the North American subspecies has not been done recently by the AOS, the AOS refers to the Birds of North America for current taxonomic treatments of subspecies (AOS, 2017, unpaginated). The Birds of North America considers *A. m. macgillivraii* and *A. m. pelonotus* to be synonymous taxa (Post & Greenlaw, 2009, unpaginated), and therefore will be considered as such for our analysis.

The currently accepted classification of MacGillivray's seaside sparrow is:
      Class: Aves
      Order: Passeriformes
      Family: Passerellidae
      Species: *Ammodramus maritimus*
      Subspecies: *Ammodramus maritimus macgillivraii*

2.2 Subspecies Description

The seaside sparrow is a drab, olive-gray (Atlantic Coast) to olive-brown bird (Gulf Coast), with a relatively long bill and short, sharp tail (see Figure 2-1 for a representative example from the Atlantic Coast). The seaside sparrow has considerable variation in size and coloration throughout its range, which has led to difficulty distinguishing subspecies. The back and breast are streaked, but it is not always conspicuous. In the Atlantic Coast subspecies, the dorsum is olive gray, with or without dark brown streaks and the underparts are grayish white, breast is heavily streaked with gray (Kale, 1996, p. 609; McDonald, 1988, p. 132). The belly is grayish white; chin and throat are white, with a dark streak extending from the jaw on the throat. The facial pattern is distinctive with a prominent yellow supraloral spot that blends to olive-green over eye (Post & Greenlaw, 2009, unpaginated).

Figure 2-1. Adult MacGillivray's seaside sparrow (*Ammodramus maritimus macgillivraii*) at Tom Yawkey Wildlife Center, Georgetown, South Carolina. Photo by Y. Laskaris, Coastal Carolina University.

MacGillivray's seaside sparrow plumage was described as darker on its backside than *A. m. maritimus*, with the back feathers and central rectrices distinctive, often broadly streaked with black, and the streaks on the chest and sides are broader and darker (Figure 2-2) (Post & Greenlaw, 2009, unpaginated; Sprunt, 1968, p. 831). Research based on the largest assemblage

of seaside sparrow skins, mostly in breeding plumage (*n* = 1,318), found that based on plumage, three of the Atlantic Coast subspecies (*A. m. maritimus, A. m. macgillivraii,* and *A. m. pelonotus*) were virtually indistinguishable (McDonald, 1988, p. 39).

All subspecies of seaside sparrow are about 14-15 centimeters (cm) (5.5-6 inches [in]) in length and males are slightly larger than females (Kale, 1996, p. 608; McDonald, 1988, p. 18). There appears to be a slight large-to-small trend from north to south, especially in the wing and tarsus measurements (McDonald, 1988, p. 19), but based on measurements, there is no significant difference between *A. m. maritimus*, *A. m. macgillivraii*, and *A. m. pelonotus* (McDonald, 1988, p. 20). In coastal Georgia, the average wing length of *A. m. macgillivraii* females was 57.3 ± 1.8 millimeters (mm) and 60.6 ± 1.8 mm for males; average *A. m. macgillivraii* female mass was 22.2 ± 1.6 grams (g) and average male mass was 22.8 ± 1.2 g (Hunter, 2016, p. 162).

Figure 2-2. The lighter, *putative Ammodramus maritimus maritimus* (left) and the darker, *putative A. m. macgillivraii* (right) captured on wintering grounds in South Carolina. Photo by S. Hartley.

The vocalization of MacGillivray's seaside sparrow has been described as peculiar, consisting of a sort of guttural roll, then a short trill, ending with a strange rasping buzz (Sprunt, 1924, p. 483). The song is often uttered in flight, the bird rising into the air, then dropping down to just above the grass tops and leveling off for varying distances, singing as it goes, before dropping suddenly into the grass. The call note might be translated as chip, but that gives little idea of the note's distinctive inflection and tonal quality. It is doubtful if the call of this subspecies can be differentiated from that of nominate *A. m. maritimus* (Sprunt, 1968, p. 834).

2.3 Current and Historical Range and Distribution

MacGillivray's seaside sparrow occupies portions of Atlantic coastal marshes in South Carolina, Georgia, and Florida in the United States. Previous MacGillivray's seaside sparrow range maps identified North Carolina as part of the subspecies' range. However, this area was excluded from our analysis, based on genetic information and a lack of suitable habitat within a portion of the area (described below in Section 2.3.1). The excluded area extends from Dare County, North Carolina to the southern boundary of Horry County, South Carolina (Figure 2-3).

The subspecies' current range extends from the northern boundary of Georgetown County, South Carolina to the St. Johns River in Duval County, Florida (Figure 2-3). The approximate length of coastline within the currently occupied range is 270 miles (435 kilometers [km]), with 150 miles (240 km) in South Carolina, 100 miles (160 km) in Georgia, and 20 miles (32 km) in Florida.

While no population extirpations or range reductions have been documented in South Carolina or Georgia to date, there has been a documented contraction of the southern range limit by approximately 100 miles (160 km) in northeast Florida (Figure 2-3; McDonald, 1988, p. 58).
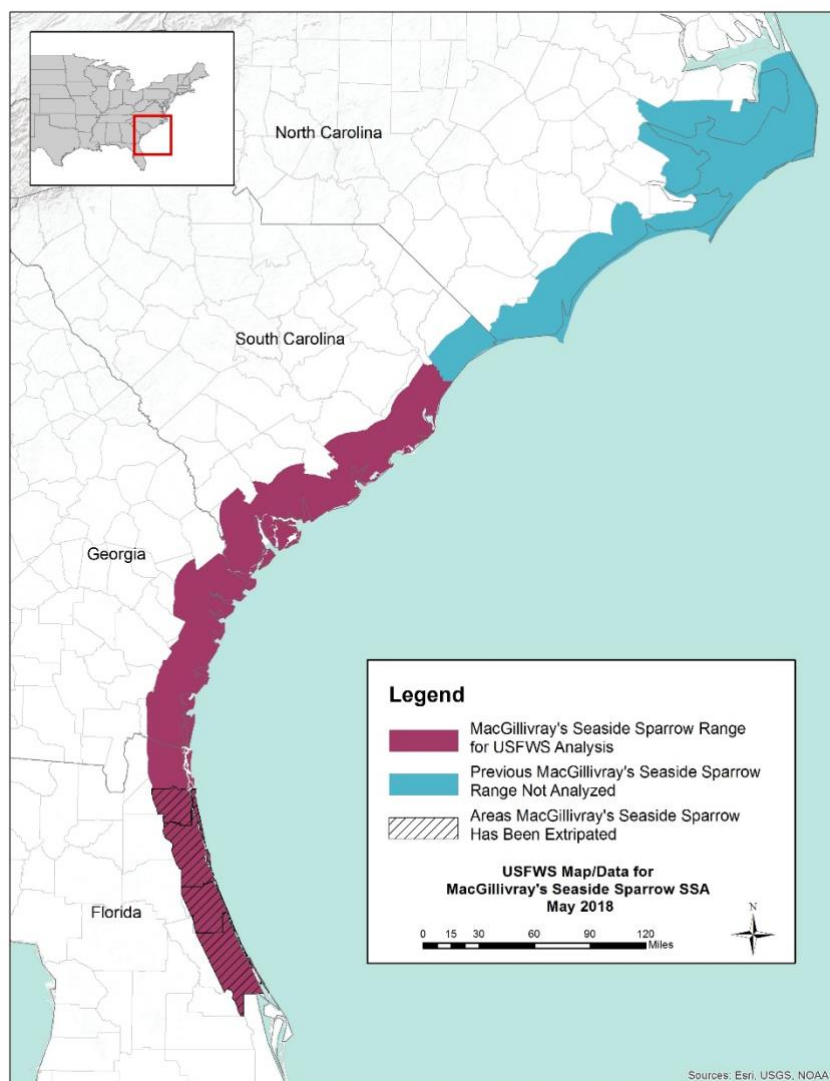
Figure 2-3. MacGillivray's seaside sparrow range map highlighting counties in purple that are considered the current range plus areas presumed to be extirpated in black hatching. Highlighted counties in blue were previously included as portions of the MacGillivray's seaside sparrow subspecies range. Recent genetic analyses support that seaside sparrows in the blue area are not genetically consistent with *Ammodramus maritimus macgillivraii* and, therefore, are excluded from this Species Status Assessment Report.

MacGillivray's seaside sparrow (= Smyrna seaside sparrow) formerly occupied marshes further south in Florida, through New Smyrna Beach (Volusia County), but the subspecies is currently presumed extirpated from these marshes. Extirpated marshes at the southern edge of the subspecies' range are discussed in Section 4.1.5 following a description of the extant breeding populations of MacGillivray's seaside sparrow.

2.3.1 Excluded Area

Recent genetic analysis supports that microsatellite genotypes of seaside sparrows in North Carolina are more consistent with northern seaside sparrows, subspecies *A. m. maritimus* (i.e., Connecticut, Delaware, Maryland, New Jersey, and Virginia; Woltmann & Hill, 2017, pp. 7, 12, 14), and not with subspecies *A. m. macgillivraii.* These preliminary genetic results showed no evidence of genetic differentiation between seaside sparrow populations near the putative contact zone between Dare County, North Carolina and Virginia (Woltmann & Hill, 2017, p. 4); Appendix A. The results of these analyses suggested no genetic differentiation between seaside sparrows in North Carolina and other northeast states.

Analysis of nine DNA samples from Yawkey Wildlife Center near Georgetown, South Carolina also suggested that the *A. m. maritimus* genotype extends even further south into the MacGillivray's seaside sparrow's range, although a lack of sufficient samples from the area precluded confirmation at the time of this assessment (Woltmann & Hill, 2017, pp. 3-4). The Yawkey Wildlife Center sampling site and the Georgia sampling site showed relatively weak differentiation from sampling locations in the subspecies *A. m. maritimus* range, but additional samples and further analysis are needed to clarify genetic differentiation outside of North Carolina (Woltmann & Hill, 2017, p. 5). The Headquarters Island site in Charleston, South Carolina, which also serves as the type locality for MacGillivray's seaside sparrow, was consistently the most differentiated population analyzed (Woltmann & Hill, 2017, p. 4).

Although seaside sparrows in North Carolina are taxonomically considered part of the *A. m. macgillivraii* subspecies published range, the best available scientific information at this time supports that the seaside sparrows in North Carolina coastal marshes are more consistent with nominate *A. m. maritimus* (Woltmann & Hill, 2017, p. 4). Genetic data suggests that a full taxonomic reassessment may be warranted (Woltmann & Hill, 2017, p. 5). Therefore, we have excluded this area from the taxonomic entity that we are reviewing in this SSA. While we recognize the difficulty in identifying a specific range boundary line on a map, the area between Georgetown, South Carolina and Southport, North Carolina has long been recognized as mostly uninhabitable to seaside sparrows and other specialist marsh birds due to a lack of well-developed salt marshes (e.g., Dingle & Sprunt Jr., 1932, p. 455 in Funderburg Jr. & Quay, 1983, p. 23; U.S. Geological Survey [USGS], 2009, unpaginated). Given the lack of suitable habitat between Georgetown, South Carolina and Southport, North Carolina and the results presented in Appendix A, we considered the northern boundary of Georgetown County, South Carolina as the northern boundary line for MacGillivray's seaside sparrow (Figure 2-3). Throughout the remainder of this SSA report, references to the range of the MacGillivray's seaside sparrow refer to the analysis area indicated in Figure 2-3.

2.4 Life History

The MacGillivray's seaside sparrow has four life stages: egg, nestling, juvenile, and adult (Figure 2-4). In the following paragraphs, we discuss each of these life stages. This information is summarized in Table 2-1, which shows the approximate timeframe for each major activity of the MacGillivray's seaside sparrow by life stage.

The egg stage begins at laying and ends at hatching, and lasts between 13-18 days depending on the clutch size. Adult females lay 1 egg per day and have an average clutch size of 3 eggs (range=1-4; Hunter, 2016, p. 163; Laskaris, 2016, p. 19; A. Schwarzer, 2017, pers. comm.). The female begins incubation with the laying of the penultimate or last egg and incubates for 11-13 days; males do not incubate (Post & Greenlaw, 2009, unpaginated). Eggs are laid in a nest cup that often has a canopy made from interweaving surrounding marsh grasses (Tomkins, 1941, p. 44) (see Section 2.5 for information about available marsh grasses). Inter-nest distances for MacGillivray's seaside sparrows range from 25 m (SD= 11m, range= 13-55m, *n*=16) at a high abundance site in South Carolina (Laskaris, 2016, p. 20) to 66 m (SE=7, range= 10-418m, *n*=89) in Florida (A. Schwarzer, 2017, unpublished data). Peak egg laying and incubation occur from late April to early July (Table 2-1). Repeated renesting following nest failure is a common

behavior in seaside sparrows (Hill & Post, 2005, p. 120; Marshall & Reinert, 1990, p. 507; Post, 1974, p. 570). For example, a MacGillivray's female in Georgia attempted to renest five times in a season (Hunter, 2016, p. 163). There also is evidence of nesting pairs having two successful nests within a season (double brooding), but this is not common (Hunter, 2016, pp. 163-164).

The nestling stage begins once an egg hatches and lasts for 9-11 days until the nestling departs from the nest and is capable of flight. Eggs hatch nearly synchronously and nestlings require full parental care at birth (Post & Greenlaw, 2009, unpaginated). Parental care is typically provided by both parents, who deliver food to the nest in mucous-bound bolus (Post & Greenlaw, 2009, unpaginated). Parents are efficient at delivering food to nestlings and there is no evidence of starvation as a source of nestling mortality for the species (Post, 1974, p. 572; Post & Greenlaw, 2006, p. 772). It takes approximately 8 days for seaside sparrow nestlings to grow from 10% to 90% of their asymptotic weight (Post & Greenlaw, 1982, p. 104). On Day 0, nestlings are in natal down, gape, and give one-note peep calls (Woolfenden, 1956, pp. 68-70). By Day 5, juvenile feathers have started to come in, and most chicks have their eyes entirely open and will actively beg for food. On Day 9, nestlings usually leave the nest within several hours of each other, but are capable of leaving the nest as early as Day 8 if disturbed by predators, stochastic events, or human disturbance (Post & Greenlaw, 2009, unpaginated; Woolfenden, 1956, p. 70). The full seaside sparrow nesting cycle, from nest building to chick fledging, takes between 24-26 days to complete and fits within the approximately 28-day window between two high lunar (spring) tides (Hunter, 2016, p. 133; Hunter, et al., 2016b, p. 136). Nestlings fledge as early as mid-April and as late as late-August (Hunter, 2016, p. 162; Tomkins, 1941, p. 45).

Nestlings start the juvenile stage once they depart the nest and are able to fly, even if the birds do not take their first true flight for a few days (Gill, 2007, p. 491). Individuals may be fed by both parents for up to 20 days following fledging (DeRagon, 1988 in Post & Greenlaw, 2009). Juveniles are believed to disperse a short distance from the nest site. In the Headquarters Island population in South Carolina, the average juvenile dispersal distance was 268 meters (m) for males (*n*=8) and 156 m for females (*n*=4; W. Post, unpublished data in Post & Greenlaw, 2009, unpaginated). In the same population, the distance between original nest site and first year breeding site was limited and similar between both sexes; the average distance between the two sites was 234 m for males (*n*=34) and 242 m for females (*n*=26; W. Post, unpublished data in Post & Greenlaw, 2009, unpaginated). In the fall, juveniles leave the breeding grounds for wintering grounds. Juveniles have been caught on wintering grounds in October in fresh fall plumage or with an active complete molt, indicating some birds complete molting during migration and some complete molting on the wintering grounds (A. Given, 2017, pers. comm.). Juveniles start a partial prebreeding molt in March of the following year on wintering sites and complete molting on the breeding grounds (Table 2-1; A. Given, 2017, pers. comm.). The juvenile stage also may be referred to as the post-fledgling or immature stage and may last up to 11 months, when an individual reaches sexual maturity.

MacGillivray's seaside sparrows reach the adult life stage the spring after hatch year when individuals become sexually mature. Adults presumably breed each year and are socially monogamous; however, there is evidence of some extra-pair paternity (offspring sired by extra-pair males) at low rates (Hill & Post, 2005, p. 119). As mentioned previously, pairs renest after nest failure and occasionally have two broods (two successful clutches) within a nesting season

(Hunter, 2016, pp. 163-164). Adults show signs of body molt in mid-March on wintering grounds and probably complete their prebreeding molt on the breeding grounds, followed by a complete postbreeding molt between July to mid-August, before leaving again for wintering grounds (Table 2-1; A. Given, 2017, unpublished data). Individuals show high site fidelity, returning to the same breeding and wintering grounds each year (Hunter, 2016, p. 163; Shaw, 2012, pp. 16-17; A. Given, 2017, unpublished data). There has been no indication of additional dispersal following the first breeding season (Post & Greenlaw, 2009, unpaginated).

Figure 2-4. MacGillivray's seaside sparrow (*Ammodramus maritimus macgillivraii*) nest with eggs (A), nestlings (B), juveniles (C), and adult (D). Photos by A. Schwarzer, Florida Fish and Wildlife Conservation Commission (B; Florida); E. Hunter, University of Georgia (A, C; Georgia); and Y. Laskaris, Coastal Carolina University (D; South Carolina).

MacGillivray's seaside sparrow is generally considered a non-migratory, sedentary subspecies, but anecdotal data and historical accounts suggest individuals exhibit local seasonal movements (between breeding and wintering seasons). In South Carolina, birds breed in marshes upriver and move to saltier coastal marshes in the wintertime (Hill & Laskaris, 2015, p. 5; Sprunt, 1926, p. 549). Birds are presumed to remain within the State and movement may even be limited to within an estuary (Hill & Post, 2005, p. 124). At present, there is one known record of a breeding bird banded at Headquarters Island, SC, and recaptured during the winter within the same estuary, 3.6 miles down the Stono River on James Island, SC (W. Post, pers. comm. in Hill & Laskaris, 2015, p. 5). Birds start appearing on the breeding grounds in mid-March (Sprunt, 1926, p. 549) and return to wintering grounds in the fall. Birds have been caught in October, but may

start arriving in September (banding on wintering grounds has not been tried in September yet; A. Given, 2017, unpublished data). In the wintering season, MacGillivray's seaside sparrow overlaps at wintering sites with the northern *A. m. maritimus* subspecies. Banding data indicates that *A. m. maritimus* individuals may outnumber *A. m. macgillivraii* individuals at high abundance wintering sites (A. Given, T. Keyes, 2017, unpublished data). However, it is important to reiterate that the plumage differences between the two subspecies are subtle and there is no known fully reliable method to distinguish between the two at this time (Hill & Laskaris, 2015, p. 5; A. Given, 2017, pers. comm.).

The lifespan of seaside sparrows is reported to be 8-9 years based on male and female captures from several populations (Post & Greenlaw, 2009, unpaginated; Sykes, 1980, p. 731). This estimate is supported by banding return data from Patuxent Wildlife Research Center which also estimates longevity to be up to 8-10 years for the seaside sparrow species (Bird Banding Laboratory, 2017, unpaginated). At least one of the three records reported is presumed to be MacGillivray's seaside sparrow; an after hatch year male was at least 10 years old during its last encounter, recaptured in June 2014 after being banded in South Carolina in March 2005 (Bird Banding Laboratory, 2017, unpaginated).

Table 2-1. The annual life cycle of MacGillivray's seaside sparrow (*Ammodramus maritimus macgillivraii*) by life stage. Lighter colored shading indicates off peak times. SM = Seasonal Movement. Sources: Ehrlich, et al., 1988, p. 570; Hunter, 2016, p. 162; Post & Greenlaw, 2009, unpaginated; Tomkins, 1941, p. 48; A. Given, C. Hill, E. Hunter, T. Schneider, A. Schwarzer, 2017, pers. comm.

Life Stage	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sept	Oct	Nov	Dec
Egg					Laying/Incubation (13-18 days)							
Nestling					Brood Rearing (9-11 days) Molt							
Juvenile			Molt							Molt Dispersal		
Adult	Wintering		Molt SM	Mating	Laying/Incubation (13-18 days) Brood Rearing (9-11 days)		Molt			Seasonal Movement Wintering		

2.5 Resource Needs (Habitat) of Individuals

MacGillivray's seaside sparrows spend their entire life in coastal salt and brackish marshes. In the northern half of the range (South Carolina), individuals breed in lower elevation areas of high marsh (marsh flooded only twice a month by spring tides) and in managed impoundments which are often brackish and non-tidal (Figure 2-5; Table 2-2; C. Hill, 2017, pers. comm.). In the winter, South Carolina birds move closer to the ocean to higher elevation areas of natural low marsh; low marsh experiences semidiurnal tides, two high and two low tides per lunar day. While South Carolina birds only winter in low marsh, birds in Georgia and Florida breed in low marsh, specifically in higher elevation areas of natural low marsh (Figure 2-6; C. Hill, E. Hunter, T. Schneider, A. Schwarzer, 2017, pers. comm.). It is uncertain where Georgia and Florida birds winter and if they make substantial seasonal movements; however, in both states, wintering seaside sparrows that are presumably *A. m. maritimus* and *A. m. macgillivraii* have been observed in shrubby marsh habitat closer to uplands (E. Hunter, A. Schwarzer, 2017, pers. comm.). MacGillivray's seaside sparrows typically avoid these areas during the breeding season and because the birds are caught in shrubs while the tide is high during winter banding efforts, it is unclear if individuals prefer shrubs during wintertime or simply require them as roost sites when the marsh is flooded (Beaton, 1993, pp. 22-23; Shaw, 2012, p. 10).

Low salt marshes across the MacGillivray's seaside sparrow range are typically dominated by smooth cordgrass (*Spartina alterniflora* [synonymous with *Sporobolus alterniflorus*]) with patches of black needlerush (*Juncus roemerianus*). High salt marsh is generally dominated by black needlerush, saltgrass (*Distichlis spicata*), and glasswort (*Salicornia virginica*). Other common salt marsh vegetation includes saltwort (*Batis maritima*), saltmeadow cordgrass (*Spartina patens*), marsh elder (*Iva frutescens*), and sea oxeye daisy (*Borrichia frutescens*). Brackish marshes are often dominated by black needlerush and big cordgrass (*Spartina cynosuroides*) with other vegetation interspersed (e.g., sand cordgrass [*Spartina bakeri*], cattails [*Typha spp.*], sedges [*Cyperus spp.*]; Hunter, et al., 2017, p. 21; Schwarzer & Cox, 2017, p. 5).

Daily, dynamic changes in water levels bring an abundance of nutrients to marsh vegetation leading to tidal marshes being one of the most productive ecosystems in the world. High primary production and marine organisms provide the terrestrial tidal-marsh faunal community, including seaside sparrows, with an abundance of food resources (Greenberg, 2006, p. 3). Seaside sparrows forage on a variety of small aquatic and terrestrial invertebrates and seeds (e.g., *Spartina*, *Salicornia*) (Sprunt, 1968, p. 833; Wilson, 1811, p. 68) in open stands of vegetation, shallow pools, pannes, and along the edges of tidal creeks (Post & Greenlaw, 2009, unpaginated) (Table 2-2). Food items are gleaned from vegetation and the substrate. Optimal habitats contain contiguous nesting and foraging areas; however, birds can successfully move between grouped nesting territories and separate feeding sites (Post, 1974, p. 572; Tomkins, 1941, p. 43).

Primary limiting factors for seaside sparrow survival and reproduction are predation, flooding, and extreme weather events (Post, 1974, p. 569; Post, 1981, p. 36; Stouffer, et al., 2013, p. 88). Breeders must mediate the trade-offs between predation risk and regular and irregular flood risk when selecting a nesting site and timing reproduction (Greenberg, et al., 2006, p. 97). As stated previously, eggs and nestlings need a nest cup that often has a canopy made from interweaving surrounding marsh grasses (Tomkins, 1941, p. 44) to provide shelter from predation. Nests are

placed along a gradient of nest height with adaptive compromises in placement to contend with the dual risks (Hunter, et al., 2016b, p. 138). Placing a nest lower to the surface in dense vegetation will increase the amount of cover to hide the nest from predators, but will increase the probability of the nest flooding (Table 2-2; Greenberg, et al., 2006, p. 97). Conversely, if a nest is placed higher in the vegetation the nest is less likely to flood, but will be more exposed to potential predators. MacGillivray's seaside sparrows avoid marshes that are near uplands, which may be an additional strategy to minimize predation risk since forests serve as sources of predators (Hunter, et al., 2017, p. 26; Picman, et al., 1993, p. 89; Schwarzer & Cox, 2017, p. 11).

Figure 2-5. Example of MacGillivray's seaside sparrow (*Ammodramus maritimus macgillivraii*) high salt marsh (left) and managed impoundment (right) habitat from Tom Yawkey Wildlife Center in South Carolina. Photo by W. Wiest, U.S. Fish and Wildlife Service.

Figure 2-6. Example of MacGillivray's seaside sparrow (*Ammodramus maritimus macgillivraii*) low salt marsh habitat from Georgia. Photo by T. Schneider, Georgia Department of Natural Resources.

Table 2-2. Resource needs (habitat) for MacGillivray's seaside sparrow (*Ammodramus maritimus macgillivraii*) to complete each life stage.

Life Stage	Resources Needs (Habitat)	References
Egg	- Lower elevation areas of high marsh, including in impoundments (SC); higher elevation areas of natural low marsh (GA/FL) - Dense herbaceous vegetative cover - Nest cup, often with canopy - Nest site above regular and extreme tide levels - Far proximity from upland	Greenberg, et al., 2006, p. 97 Hunter, et al., 2016b, p. 138 Hunter, et al., 2017, p. 26 Post & Greenlaw, 2009, unpaginated Schwarzer & Cox, 2017, p. 11 Tomkins, 1941, p. 44 C. Hill, E. Hunter, T. Schneider, A. Schwarzer, 2017, pers. comm.
Nestling	- Lower elevation areas of high marsh, including in impoundments (SC); higher elevation areas of natural low marsh (GA/FL) - Dense herbaceous vegetative cover - Nest cup, often with canopy - Nest site above regular and extreme tide levels - Far proximity from upland - Food – aquatic/terrestrial invertebrates, seeds	Greenberg, et al., 2006, p. 97 Hunter, et al., 2016b, p. 138 Hunter, et al., 2017, p. 26 Post & Greenlaw, 2006, p. 765 Post & Greenlaw, 2009, unpaginated Schwarzer & Cox, 2017, p. 11 Tomkins, 1941, p. 44 C. Hill, E. Hunter, T. Schneider, A. Schwarzer, 2017, pers. comm.
Juvenile / Adult	- Lower elevation areas of high marsh, including in impoundments (SC); higher elevation areas of natural low marsh (GA/FL; SC - wintering) - Dense herbaceous vegetative cover that allows movement underneath - Food – aquatic/terrestrial invertebrates, seeds - Openings in vegetation (e.g., pools, pannes, and edges of tidal creeks) - High tide roosting sites in the marsh	Beaton, 1993, pp. 22-23 Greenberg, et al., 2006, p. 97 Post & Greenlaw, 2006, p. 765 Post & Greenlaw, 2009, unpaginated Shaw, 2012, p. 10 C. Hill, E. Hunter, T. Schneider, A. Schwarzer, 2017, pers. comm.

CHAPTER 3 – FACTORS INFLUENCING VIABILITY

The following discussion provides a summary of the factors that are affecting or could be affecting the current and future condition of the MacGillivray's seaside sparrow throughout some or all of its range. Those risks that are not known to have effects on MacGillivray's seaside sparrow populations, such as overutilization for commercial and scientific purposes and disease, are not analyzed in this SSA report. In addition, there are other factors, such as fire, pollution, and oil spills, which may have some localized impacts on individual MacGillivray's seaside sparrows but are not currently known to be serious factors affecting viability of the subspecies, nor are anticipated to have population- or subspecies- level impacts in the future.

3.1 Predation

Predation is a significant mortality factor for MacGillivray's seaside sparrow nests (Hunter, et al., 2016b, p. 138; Schwarzer & Cox, 2017, p. 11), as well as for other Atlantic Coast seaside sparrow subspecies (Kern, et al., 2012, p. 935; Post, et al., 1983, p. 132; Roberts, et al., 2017, pp. 123-127). Although nest predation affects all seaside sparrow populations, it likely occurs more frequently at lower latitudes, due to the general pattern of increasing mammal abundance with decreasing latitude. Predation accounts for the majority of MacGillivray's seaside sparrow nest losses, followed by tidal flooding (Table 4-3). In recent demographic studies of MacGillivray's seaside sparrow, predation was the primary source of nest mortality with 28%, 63%, and 64% of known-fate nests lost to predators in South Carolina, Georgia, and Florida, respectively (Table 4-3; Laskaris, 2016, p. 66; E. Hunter, A. Schwarzer, 2017, unpublished data). MacGillivray's seaside sparrow nest predation is related to nest height and nest cover (Hunter, et al., 2016b, p. 138). Seaside sparrows face a trade-off between nest predation and nest flooding in terms of nest site selection. Nests placed higher from the marsh surface may be more vulnerable to predation, while nests placed closer to the ground may be more vulnerable to flooding (Greenberg, et al., 2006, p. 97; Hunter, et al., 2016b, p. 138). Furthermore, MacGillivray's seaside sparrows may alter their nest site selection behavior in response to perceived predation risk (Hunter, et al., 2016b, pp. 139-141).

Predators of seaside sparrow nests include raccoons (*Procyon lotor*), American mink (*Neovison vison*), rice rats (*Oryzomys palustris*), fish crows (*Corvus ossifragus*), red-winged blackbirds (*Agelaius phoeniceus*), boat-tailed grackles (*Quiscalus major*), marsh wrens (*Cistothorus palustris*), northern harriers (*Circus cyaneus*), and snakes (Hunter, et al., 2016b, p. 136; Post, et al., 1983, p. 134; Post & Greenlaw, 2009; Schwarzer & Cox, 2017, pp. 13-14). However, more study is needed to verify predator identity and to determine the relative impact of individual predator species throughout the seaside sparrow range (Roberts, et al., 2017, p. 128). The most common nest predators that nest sympatrically in marshes with seaside sparrows are rice rats, red-winged blackbirds, and marsh wrens. Uplands and wooded areas that border tidal marshes are sources of additional nest predators, although distance to forest had no effect on predation rate in Georgia (Hunter, et al., 2016b, pp. 136-139) and only a slight negative effect on nest survival in Florida (A. Schwarzer, 2018, unpublished data). However, this lack of a strong relationship may be due in part to MacGillivray's seaside sparrows avoiding marshes near forests in general (Hunter, et al., 2017, pp. 25-26).

The primary predators of individual seaside sparrows (juveniles and adults) are avian species, including northern harrier, American crow (*Corvus brachyrhynchos*), and owls (Post & Greenlaw, 2009; Woolfenden, 1956, pp. 50-51). While little is known about predation rates on juveniles, predation on adults during the breeding season appears to be uncommon (Post & Greenlaw, 2009, unpaginated). Adult male and female seaside sparrows experience relatively higher adult survival rates during the breeding season compared to the wintering season (Borowske, 2015, pp. 103-104).

3.2 Tidal Flooding

Tidal flooding is a primary abiotic driver of marsh ecology. Daily and monthly lunar cycles result in periodic tidal flooding of coastal areas. Lunar-driven tidal flooding can interact with or be amplified by stochastic events, such as high wind, storms, and rainfall. Seaside sparrows are affected by tidal flooding during all stages of their life history, although the greatest demographic impact is on nests. At a marsh in Georgia, 17% of MacGillivray's seaside sparrow nests were lost to flooding from 2013-2015 (E. Hunter, 2017, unpublished data), and 6% of nests were lost to flooding at a marsh in Florida from 2015-2017 (Table 4-3; A. Schwarzer, 2017, unpublished data). Despite the low number of direct nest mortalities from flooding, larger tidal amplitudes of high tides had a strong negative effect on daily nest survival probability in Florida (A. Schwarzer, 2018, unpublished data), indicating that there may be secondary impacts from tidal flooding such as increased concentrations of predators in breeding areas during the highest high tides. No MacGillivray's seaside sparrow nests were flooded at a marsh in South Carolina in 2015; however, the absence of nest failure due to inundation may have been due to restrictions in tidal flow at two of the three study sites and a small sample size (Laskaris, 2016, pp. 13-14, 19). While tidal flooding is an important cause of nest failure for seaside sparrows, in addition to predation, the relative importance of tidal flooding versus predation is highly variable from year to year (Hunter, et al., 2016b, p. 138; Marshall & Reinert, 1990, p. 507; Roberts, et al., 2017, p. 123).

Tidal marsh-nesting birds have developed adaptations that reduce nest losses to tidal flooding, such as rapidly renesting after nest failure so that the nest cycle fits between lunar tidal peaks, which occur at approximately 28-day intervals (Reinert, 2006, pp. 88-89; Shriver, et al., 2007, pp. 555-557). Seaside sparrow nestlings have also been observed climbing above their nests during flood events to avoid being drowned (Marshall & Reinert, 1990, p. 508). Elevating nests above the marsh surface is another strategy used to reduce nest flooding (Greenberg, et al., 2006, p. 97), but it does not always result in higher nest success. These adaptations do not eliminate all nest losses to tidal flooding, due to the unpredictability of tidal amplitude and trade-offs with nest predation. Although MacGillivray's seaside sparrows sometimes modify their nest site selection following a nest failure by flooding, the overall unpredictability of tidal amplitude means that some failures will still occur (Hunter, et al., 2016b, pp. 138-141). Furthermore, as described in Sections 2.5 and 3.1, seaside sparrows face a trade-off in nest site selection based on the competing risks of nest predation and nest flooding. Placing nests higher from the ground to avoid flooding may increase the risk of nest predation due to lack of cover (Greenberg, et al., 2006, p. 97; Hunter, et al., 2016b, p. 138). Sea-level rise is predicted to change the tidal regime by increasing the frequency and severity of tidal flooding, and MacGillivray's seaside sparrows may have very low rates of nest success under extreme sea-level rise, especially when combined

with high nest predation (Hunter, 2017, pp. 464-466). Management actions intended to reduce the impacts of more frequent and severe flooding are logistically complex and may be infeasible. However, actions to reduce predation risk may have the simultaneous benefit of reducing nest flooding because predation and flood risk are linked through nest-site selection and adaptive behaviors (Hunter, 2017, p. 467). See Section 3.3 for further discussion on sea-level rise (and coastal storms).

Extreme inundation caused by storms is not thought to commonly cause adult mortality, since adult seaside sparrows are able to find refuge on marsh vegetation that protrudes above the water during extreme flooding (Post & Greenlaw, 2009, unpaginated). However, sufficiently tall, available marsh vegetation can be a limited resource during extreme events. During one such event of 24 hours of rain and an unusually high tide in Georgia during the wintering season, 228 seaside sparrows (with eight saltmarsh sparrows and six marsh wrens) were observed attempting to perch on a single patch of exposed marsh grass approximately 8m-long and 1m-wide (Beaton, 1993, pp. 22-23). Space appeared to be severely limited and individuals were observed scrambling over one another to find a suitable perch. If tall marsh vegetation is a limiting resource during extreme inundation events, tidal flooding may be a greater factor for adult mortality.

3.3 Climate Change

Tropical storms and hurricanes are projected to increase in frequency and size, including in the North Atlantic Basin (Knutson, et al., 2015, p. 7221), simultaneously with increases in mean sea-level (Sweet, et al., 2017, entire) as consequences of global climate change. Sea-level rise and storm surge from extreme climatic events lead to higher tides and consequently increased flood risk to nesting marsh birds. Extreme flooding events are exacerbated by habitat change, and increases in catastrophic flooding in terms of tide height and flooding frequency lead to reproductive failure (i.e., nest loss; Bayard & Elphick, 2011, pp. 398-400; Hunter, 2017, pp. 464-465). A storm coinciding with the breeding season could severely decrease reproductive success in a given year (van de Pol, et al., 2010, p. 727). As mentioned in Section 3.2, MacGillivray's seaside sparrows nest in an approximately 26-day window in between the lunar tidal peaks, which occur at approximately 28-day intervals (Hunter, 2017, pp. 460-462). As sea-level rise continues, the 26-day nesting window will shrink because the high spring tides that flank either side of the window are projected to last longer after the most recent spring tide and arrive sooner in advance of the next month's spring tide (Field, et al., 2016a, p. 2059). Higher tides caused by coastal storms and sea-level rise effectively diminish the ability for MacGillivray's seaside sparrows to fledge chicks within the window necessary for mating, laying, incubation, and the nestling stage; therefore, it is unsurprising that MacGillivray's seaside sparrows may have very low rates of nest success under extreme sea-level rise conditions (Hunter, 2017, pp. 464-466).

In the short term, tidal flooding of individual nests from storm surge and sea-level rise is likely to be the greatest factor influencing viability of MacGillivray's seaside sparrow and other marsh birds that is attributable to climate change. This factor is currently driving population declines in a sister-taxon to MacGillivray's seaside sparrow, the saltmarsh sparrow (*Ammodramus caudacutus*; Field, et al., 2016a, pp. 2065-2066). However, understanding of how higher elevation areas of salt marsh can be maintained in light of rising relative sea-levels (either

through slowing the rate of change or by facilitating marsh migration) will be essential to maintaining tidal marsh bird populations in the long term. Sea-level rise, estimated to be 50-200 cm over the next century, is predicted to cause a loss of wetland and terrestrial habitat along coastlines worldwide (Mitsch & Gosselink, 2015, p. 579). While tidal marsh habitat can gain surface elevation through accretion, there is growing concern that the rate of sea-level rise will outpace marsh accretion in many locations, resulting in a loss of marshes due to inundation, erosion, and saltwater intrusion (Mitsch & Gosselink, 2015, p. 579). Historically, tidal marshes have adapted to sea-level rise by moving inland; however, many marshes will be prevented from moving inland by human development, dikes or bulkheads, or by the steep slope of the adjacent uplands (Mitsch & Gosselink, 2015, pp. 579-580). Marshes may be converted into open water or may simply become unsuitable for vegetation growth, resulting in open mud flats. Although total marsh area appears to no longer be declining rapidly, the nature of the habitat continues to change, with a shift toward wetter conditions and increases in the amount of low marsh as indicated by changing vegetation (for an example see Field, et al., 2016b, pp. 365-366). These changes in marsh morphology and vegetation are expected to have negative impacts on many tidal marsh-nesting species (Erwin, et al., 2006, pp. 223-224).

The predicted habitat changes due to sea-level rise match those predicted for natural marshes inhabited by MacGillivray's seaside sparrow (e.g., in Georgia [Hunter, et al., 2017, pp. 25-27] and Florida [Hagen, et al., 2013, entire]). The nesting and foraging niches of MacGillivray's seaside sparrow in Georgia have been predicted to decrease in area, patch size, and permanence with sea-level rise (Hunter, et al., 2015, p. 1533). Under a 1-m increase in sea-level by 2100, MacGillivray's seaside sparrow high abundance habitat in Georgia may decrease by 81%, resulting in a decrease in population size (Hunter, et al., 2017, pp. 25, 27); this is further discussed in Chapter 5 – Future Condition. This decline is predicted to begin around 2025-2050 and continue through 2100 (Hunter, et al., 2017, pp. 25-26). Notwithstanding uncertainties in the extent of future marsh habitat, MacGillivray's seaside sparrow abundance is very likely to decrease, because it is highly sensitive to changes in relative surface elevation, which is directly related to sea level (Hunter, et al., 2017, p. 26).

3.4 Habitat Conversion and Degradation

3.4.1 Development

Coastal development can have numerous negative impacts on tidal marshes and endemic species, including MacGillivray's seaside sparrow. The most significant and immediate impact to tidal marshes is the direct conversion of marsh to a developed area and consequently the loss of habitat to tidal marsh inhabiting species. In addition, development can have indirect effects on marsh vegetation. When development involves the removal of vegetation bordering the upland edge of marshes, run-off into marshes may be increased. This run-off not only decreases marsh soil salinity, but can also bring nutrients into the system, such as nitrogen. Freshwater and nutrient-rich run-off can cause shifts in tidal marsh plant communities (Bertness, Silliman, & Holdredge, 2009, p. 141) and can result in increased cover of invasive plants, such as the common reed (*Phragmites australis*), which is not suitable nesting habitat for MacGillivray's seaside sparrow (Silliman & Bertness, 2004, p. 1430).

Barrier islands, behind which large expanses of tidal marsh habitat naturally occurs (Bahr & Lanier, 1981, p. 3; Beccasio, et al., 1980, p. 114), traditionally have contributed (via storm protection and the dispersal of sediment) to the larger ecosystem via natural coastal processes. Today however, many of these barrier islands have been developed and can no longer function in their natural capacity due to jetties, beach groins, and beach re-nourishment resulting in abnormal accretion in some areas and severe erosion on other islands (Hayes, et al. 1975, pp. G 15-163). The construction of causeways to access the islands also disrupts the natural marsh by disrupting tidal flows and further fragmenting tidal habitat.

MacGillivray's seaside sparrows currently occupy marshes that are relatively distant from development and forested areas (Hunter, et al., 2016a, p. 5); however, because future sea-level rise is expected to "push" available habitat further inland (closer to developed areas), MacGillivray's seaside sparrows will likely be forced to occupy habitat closer to development, or alternatively may simply not occupy it and the habitat will be lost. Coastal tidal marshes are threatened by the "coastal squeeze", the combination of sea-level rise and a physical barrier that prevents the landward migration of marshes (Martinez, et al., 2014, p. 180); this barrier is often man-made or at least a result of human development, such as shoreline armoring, roads, or other infrastructure. As a result of this coastal squeeze, the quality of the remaining marsh habitat is likely to decline, since developed areas are sources of toxins and predators of MacGillivray's seaside sparrow (Hunter, et al., 2017, p. 23). In some cases, marsh habitats may be lost completely unless shoreline protection structures (e.g., seawalls, bulkheads) or other barriers are removed and marshes have the opportunity to migrate landward (Roman, 2017, p. 714).

3.4.2 Mangrove Encroachment

Tidal salt marshes become unsuitable for MacGillivray's seaside sparrow when they are invaded by woody vegetation. MacGillivray's seaside sparrow can tolerate the early invasion of salt marshes by mangroves, but will abandon the site when mangroves become established in a large portion of the area (Kale, 1996, p. 611). In northeast Florida, red (*Rhizophora mangle*) and black mangroves (*Avicennia germinans*) have been expanding their range northward due to a general increase in temperature (Cavanaugh, et al., 2014, p. 724). The historical (1942-1980) northern limit of mangroves on the U.S. Atlantic coast, presumably dictated by cold temperatures, was approximately 30°N, just north of St. Augustine, FL (Cavanaugh, et al., 2014, p. 723; Rodriguez, et al., 2016, p. 246). Whereas the southern limit of temperate salt marshes was approximately 28°N, and mangroves and salt marshes coexisted between the two latitudes (28°N to 30°N; Cavanaugh, et al., 2014, p. 723). The northern half of the historical mangrove-salt marsh ecotone used to support MacGillivray's seaside sparrows, but with mangrove expansion, the subspecies was extirpated from the most southern extent of its range (approximately 29.0°N to 30.4°N; Kale, 1983, pp. 42-45).

In the 1920s, mangroves were present on the fringes and as scattered trees in the marsh habitat occupied by MacGillivray's seaside sparrow in New Smyrna, FL (Nicholson 1928 in Kale 1983, p. 44). By the mid-1940s, the mangroves had increased considerably in area and size, and MacGillivray's seaside sparrow abundance had decreased (Nicholson 1946, 1950 in Kale, 1983, p. 44). Mangrove encroachment gradually led birds to abandon previously occupied habitat, leading to the extirpation of the subspecies from New Smyrna, FL to the St. Johns River near

Jacksonville, FL (Kale, 1983, pp. 42-45; McDonald, 1988, p. 55). The northward expansion of mangroves in northeast Florida carries on today, as evidenced by the doubling of the spatial extent of mangroves between 29° and 29.75°N from 1984 to 2011 (Cavanaugh, et al., 2014, p. 724). Because general increases in temperature and decreases in the frequency of extreme cold events are expected to continue with global climate change (Cavanaugh, et al., 2014, p. 723), black mangroves within the MacGillivray's seaside sparrow range are projected to expand 2 degrees latitude toward the North and South Poles by 2080 (using a 3 m sea-level rise scenario; Record, et al., 2013, pp. 11-12). Given the predictions that mangrove expansion will continue and the propensity of MacGillivray's seaside sparrows to abandon marshes invaded by mangroves, it is plausible that the current MacGillivray's seaside sparrow southern range will contract further in the future. See Sections 2.3 and 4.1.5 for additional information on the current and historical range and areas presumed extirpated.

3.4.3 Ditching and Impoundments

Much of the natural tidally-influenced brackish and salt marshes that MacGillivray's seaside sparrows inhabit along the South Atlantic Coast have been impacted in some way by anthropogenic activities. In some areas, the impacts have been relatively minor, whereas in other areas habitat degradation has been more significant resulting in a complete loss of the values and functions of the natural marsh. Along the Atlantic Coast, large areas of tidal marsh have been ditched and/or impounded for mosquito control (O'Bryan & Carlson, 1995, pp. 99-101; Whitman, 1995b, p. A 45; Widjeskop, 1995, pp. A 40-41;). In some areas this resulted in dewatered marsh, in other areas the marsh is flooded during extended periods during the summer. In general, mosquito control has resulted in an altered hydrology (Clarke, et al., 1984, pp. 177-178), conversion of tidal vegetated marsh, introduction of exotic plants (e.g., *Phragmites australis* in more northern latitudes), and habitat fragmentation (Baldassarre & Bolen, 1994, p. 417; Whitman, 1995a, p. E 42). Impounding of marshes for mosquito control and waterfowl production was the primary cause of extirpation and eventually extinction in the related dusky seaside sparrow (*A. m. nigrescens;* Kale, 1996, p. 612).

In the MacGillivray's seaside sparrow range in South Carolina (and to a much lesser extent in Georgia), an even greater impact to the natural marsh than mosquito control was the construction of rice field impoundments. During the 18th and 19th centuries, tens of thousands of acres of marsh and forested wetlands were cleared and diked for the cultivation of rice. When the rice culture collapsed, many of these impoundments were maintained for waterfowl hunting (Strange, 1995a, p. D 1; Strange, 1995b, p. A 55). Later in the 20th century, many additional brackish and salt marshes were impounded, also for waterfowl hunting and/or for mosquito control (Ferrigno, 1995a, pp. D 16-18; Johnson, 1995, p. A 58; Meredith, et al., 1995, pp. D 2-7; Monschein, 1995, p. A 53; Wiegart & Freeman, 1990, p. 63; Williams, et al., 2002, p. 10). Tidal impoundments typically have perimeter dikes that extend several feet above the bed of marsh. Adjacent to the perimeter dikes are large perimeter ditches that were excavated for fill material for the dikes that also circulate tide water throughout the impoundment. Within the impoundment, a series of shallower ditches were also excavated to further facilitate water flow. Many larger impoundments also have multiple cross-dikes that break up larger areas of the marsh bed and allow for greater hydrologic control in more compact, compartmentalized units. The hydrology, other than rainfall or reserves, is typically controlled through a series of water control structures

which in the Southeast is based on the historic rice field trunks (Williams, 1995, pp. D 30-55; Williams, et al., 2002, p. 10).

Tidal impoundments are managed with water level manipulations and disturbance to control the succession of vegetation. This disturbance can mimic natural events (e.g., prescribed fire) or can be more invasive such as mechanical disking or the application of herbicides. Generally the goal is to create a 50:50 dispersion of open water and emergent vegetation (Perry, 1995, p. D 126). The manipulation of water within the impoundment is very artificial relative to the natural, twice daily tide cycle. Water levels vary seasonally and are often maintained at static levels. During the growing season they typically are either drawn down for moist soil management or are flooded to control nuisance vegetation or managed for specific plant species (e.g., widgeon grass) (Ferrigno, 1995b, p. D 121; Prevost, 1995, pp. C 12-17). This results in habitat that is generally unsuitable for meeting MacGillivray's seaside sparrow habitat requirements (Table 2-2). However, while most impoundments contain altered habitat that is unusable by MacGillivray's seaside sparrows, the subspecies does currently occupy impounded marshes at specific sites in South Carolina. These occupied impoundments are not specifically managed for MacGillivray's seaside sparrows, but because of the topographic variability within the impoundments, suitable marsh habitat remains after the impoundments have been flooded. Many land managers today have a multi-species approach and attempt to provide habitat based on the seasonal requirements of different groups of birds (e.g., sheet water habitat for migrating shorebirds). Current management practices for impoundments occupied by MacGillivray's seaside sparrow inadvertently maintain suitable habitat for the subspecies that would otherwise succeed into unsuitable shrub-dominated habitat. Overall, historic ditching and impoundments likely have had a larger impact on MacGillivray's seaside sparrows in South Carolina than in Georgia and Florida, owing to these activities occurring more frequently and to a greater degree in brackish marshes where MacGillivray's seaside sparrows in South Carolina often nest.

3.4.4 Dredging and Harbor Deepening

Many tidal marshes in the Southeast also have been altered through dredging and harbor deepening projects. Dredging can be small scale, such as the removal of sediment from a tidal creek, or large scale, such as maintenance dredging for the Atlantic Intracoastal Waterway or the deepening of harbors for commercial shipping traffic (Tiner, 1993, p. 2). The construction of the Atlantic Intracoastal Waterway itself had a significant impact on the natural marsh through the conversion of tidal marsh to a subtidal ditch as well as the alteration of the hydrology of the surrounding marsh. The construction of the Atlantic Intracoastal Waterway and the dredging of waterways and harbors also necessitated the need for spoil disposal areas (Gill, et al., 1995, p. E 65; Knoch, 1995, p. E 34). Subsequently thousands of acres of potential MacGillivray's seaside sparrow habitat have been lost as tidal marsh was diked and filled with spoil material. Deepening harbors in order to meet the requirements of larger commercial container ships can also result in a significant change in the salinity of surrounding estuaries. Harbor deepening projects and maintenance dredging are ongoing in the MacGillivray's seaside sparrow range (e.g., Charleston Harbor Post 45 Deepening Project, Savannah Harbor Expansion Project, and Jacksonville Harbor Deepening Project) and these anthropogenic activities affect the viability and adaptability of the tidal marsh, an environment already exposed to dynamic natural coastal processes (Dahl & Steadman, 2013, p. 6). However, while individual MacGillivray's seaside sparrows may occur

within or adjacent to the footprint of these harbor deepening and dredging projects, the projects are primarily concentrated in areas outside of the marshes supporting current breeding populations of the subspecies.

3.5 Conservation Measures

3.5.1 Migratory Bird Treaty Act

The Migratory Bird Treaty Act of 1918 (16 U.S.C. 703 *et seq.*) is the Federal law providing specific protection for the MacGillivray's seaside sparrow due to the species' status as a migratory bird. The Migratory Bird Treaty Act (MBTA) prohibits the following actions, unless permitted by Federal regulation: to "pursue, hunt, take, capture, kill, attempt to take, capture or kill, possess, offer for sale, sell, offer to purchase, purchase, deliver for shipment, ship, cause to be shipped, deliver for transportation, transport, cause to be transported, carry, or cause to be carried by any means whatever, receive for shipment, transportation or carriage, or export, at any time, or in any manner, any migratory bird…or any part, nest, or egg of any such bird." Through issuance of Migratory Bird Scientific Collecting permits, the Service ensures that best practices are implemented for the careful capture and handling of MacGillivray's seaside sparrows during banding operations and other research activities. The December 22, 2017 Solicitor's Opinion, Opinion M-37050, concludes that "consistent with the text, history, and purpose of the MBTA, the statute's prohibitions on pursuing, hunting, taking, capturing, killing, or attempting to do the same apply only to affirmative actions that have as their purpose the taking or killing of migratory birds, their nests, or their eggs." Therefore, take of a MacGillivray's seaside sparrow, its chicks, or its eggs that is incidental to another lawful activity does not violate the MBTA.

3.5.2 Coastal Management

The Coastal Zone Management Act of 1972 (P.L. 92-583) (86 Stat. 1280; 16 U.S.C. 1451-1464) provides Federal funding to implement the states' federally approved Coastal Zone Management Plans. All states in the MacGillivray's seaside sparrow's range have approved Coastal Zone Management Plans, which guide and regulate development and other activities within the designated coastal zone of each state (National Oceanic and Atmospheric Administration [NOAA], 2016, unpaginated). The Federal Consistency provision of the Coastal Zone Management Act requires Federal action agencies to ensure that the activities they fund or authorize are consistent, to the maximum extent practicable, with the enforceable policies of that state's federally approved coastal management program (16 U.S.C. 1456).

The Clean Water Act (i.e., Federal Water Pollution Control Act) (33 U.S.C. 1251 *et seq.*)) and the Rivers and Harbors Act have sections (404 and 10, respectively) that contain provisions for the protection of jurisdictional wetlands from filling activities. The U.S. Army Corps of Engineers in conjunction with the U. S. EPA administers permits that consider avoidance, minimization and compensation for projects affecting wetlands. Projects that cannot avoid impacts to wetlands must mitigate their impacts through a restoration action for the equivalent functional loss. Mitigation banks are often used which tend to centralize restorative actions at a specific location for impacts in a considerably wider service area. Exact wetland types that are negatively impacted are not always restored.

3.5.3 Conservation Lands

Suitable habitat for MacGillivray's seaside sparrow can be found within National Wildlife Refuges (NWR), Cumberland Island National Seashore, preserves, WMAs, and other conservation lands across the subspecies' range (Table 3-1). The National Wildlife Refuge System Improvement Act of 1997 (16 U.S.C. 668dd *et seq*.) (NWRSIA) establishes the protection of biodiversity as the primary purpose of the NWR system; recreational and other uses of a NWR may only be approved if the Service finds such uses to be compatible with the purposes of that individual NWR and the purposes of the NWR system.

Habitat for MacGillivray's seaside sparrow also can be found within Cumberland Island National Seashore and Timucuan Ecological and Historic Preserve (Table 3-1), which must balance visitation and recreation with the protection of natural resources like the MacGillivray's seaside sparrow and its habitat. The National Park Service Organic Act of 1916, as amended (39 Stat. 535, 16 U.S.C. 1) (NPSOA), states that the National Park Service (NPS) "shall promote and regulate the use of [NPS units]…to conserve the scenery and the national and historic objects and the wildlife therein and to provide for the enjoyment of the same in such manner and by such means as will leave them unimpaired for the enjoyment of future generations." In addition to the NPSOA, the MacGillivray's seaside sparrow may benefit from a 2010 nonregulatory Memorandum of Understanding (MOU) between the NPS and the Service regarding migratory birds that was executed pursuant to Executive Order 13186; section F.4. of the MOU states that the NPS will identify and protect natural habitats of migratory bird species within park boundaries.

In addition to federal lands, there are other protected lands that are managed by state, private, and non-governmental organizations occurring within the range of the MacGillivray's seaside sparrow (Table 3-1).

However, although MacGillivray's seaside sparrow occurs in suitable habitat on conservations lands, the primary threats of predation and tidal flooding are not ameliorated by land protection, unless specific management is conducted to reduce these threats. Therefore, these primary threats to the MacGillivray's seaside sparrow would still pose a risk to the future viability of the subspecies even when suitable habitat occurs on conservation lands.

Table 3-1. Major protected conservation areas within the current range of MacGillivray's seaside sparrow (*Ammodramus maritimus macgillivraii*). Acreages based on the Protected Areas Database (U.S. Geological Survey, Gap Analysis Program [USGS GAP], 2016, unpaginated). Abbreviations: NGO= non-governmental organization, WMA= wildlife management area.

State	Manager	Protected Area	Acres (Hectares)
South Carolina	Federal	Cape Romain National Wildlife Refuge	64,674 (26,173)
		Ernest F. Hollings Ace Basin National Wildlife Refuge	12,046 (4,875)
		Pinckney Island National Wildlife Refuge	4,044 (1,637)
		Savannah National Wildlife Refuge	15,021 (6,079)
	NGO	Red Bluff Marsh Preserve	1,797 (727)
	State	Bear Island WMA	12,056 (4,879)
		Belle W. Baruch Institute for Marine & Coastal Sciences	15,955 (6,457)
		Botany Bay Plantation Heritage Preserve / WMA	3,599 (1,456)
		Capers Island Heritage Preserve	2,171 (879)
		Daws Island Heritage Preserve	3,028 (1,225)
		Morgan Island	4,969 (2,011)
		Old Island Heritage Preserve / WMA	3,004 (1,216)
		Saint Helena Sound Heritage Preserve / WMA	18,453 (7,468)
		Santee Coastal Reserve WMA	25,766 (10,427)
		Santee-Delta WMA	1,822 (737)
		Tom Yawkey Wildlife Center Heritage Preserve	19,761 (7,997)
		Turtle Island WMA	1,646 (666)
		Victoria Bluff Heritage Preserve / WMA	1,069 (433)
Georgia	Federal	Blackbeard Island National Wildlife Refuge	8,333 (3,372)
		Cumberland Island National Seashore	45,905 (18,577)
		Fort Pulaski National Monument	5,132 (2,077)
		Harris Neck National Wildlife Refuge	2,754 (1,115)
		Savannah National Wildlife Refuge	13,312 (5,387)
		Wassaw National Wildlife Refuge	10,228 (4,139)
		Wolf Island National Wildlife Refuge	9,718 (3,933)
	NGO	Driftwood Plantation	3,781 (1,530)
	Private	St. Catherine's Island	14,903 (6,031)
	State	Altamaha WMA	29,264 (11,843)
		Hofwyl-Broadfield Plantation Historic Site	1,265 (512)
		Little Tybee-Cabbage Island Natural Area	8,223 (3,328)
		Ossabaw Island WMA	28,200 (11,412)
		Richmond Hill WMA	25,796 (10,439)
		Sapelo Island WMA	18,459 (7,470)
		Wormsloe State Historical Site	1,993 (807)
Florida	Federal	Timucuan Ecological and Historic Preserve	45,695 (18,492)
	NGO	Machaba Balu Preserve	2,146 (868)
	State	Big Talbot Island State Park	1,669 (675)
		Fort Clinch State Park	2,191 (887)
		Little Talbot Island State Park	1,794 (726)
		Pumpkin Hill Creek Preserve State Park	4,078 (1,650)

3.5.4 State Protections

MacGillivray's seaside sparrow is not listed as state threatened or endangered for any of the states within the subspecies' range (Table 3-2). In Florida, MacGillivray's seaside sparrow is ranked as a species of greatest conservation need (Florida Fish and Wildlife Conservation Commission [FWC], 2012, p. 57) and in Georgia, it is ranked as a high priority (Georgia Department of Natural Resources [GADNR], 2015, p. A-3). In South Carolina, it is listed as a moderate priority species (South Carolina Department of Natural Resources [SCDNR], 2015, p. A-1).

Table 3-2. MacGillivray's seaside sparrow (*Ammodramus maritimus macgillivraii*) listing status and rank by state (north to south) within the range of the subspecies.

State	State Listing	State Rank/Priority
South Carolina	Not Listed	Moderate
Georgia	Not Listed	High
Florida	Not Listed	Species of Greatest Conservation Need

3.5.5 Summary of Conservation Actions

The Migratory Bird Treaty Act protects the MacGillivray's seaside sparrow from direct take and purposeful actions. Some laws do regulate discharges into wetlands and coastal development, including section 404 of the Clean Water Act and the Coastal Zone Management Act, as implemented by Federal and state regulations. The National Park Service Organic Act and the National Wildlife Refuge System Improvement Act provide protection for the MacGillivray's seaside sparrow from habitat loss and inappropriate management on Federal lands.

CHAPTER 4 – POPULATION AND SUBSPECIES NEEDS AND CURRENT CONDITION

In this chapter, we consider the MacGillivray's seaside sparrow's historical distribution, its current distribution, and what the subspecies needs for viability. We first defined populations of the subspecies. Next, we characterize the needs of the subspecies in terms of population resiliency and subspecies' representation and redundancy (the 3Rs). Finally, we estimate the current condition of MacGillivray's seaside sparrow using population and habitat metrics used to characterize the 3Rs.

4.1 Populations

In this assessment, we defined a population of MacGillivray's seaside sparrow as a group of inter-breeding individuals that live in the same place at the same time. Birds in South Carolina are widespread, but exhibit a scattered and patchy distribution where birds are locally abundant at just a few sites, or occur in substantially lower densities or not at all outside of the high abundance areas (Appendix B; Hill & Laskaris, 2015, pp. 2-3). In Georgia and northeast Florida, MacGillivray's seaside sparrows also are widespread, but are distributed more uniformly along the states' coastlines (Hunter, et al., 2016a, pp. 4-6; Schwarzer & Cox, 2017, p. 2). Based on the subspecies' distribution pattern from survey data and the connectivity of marsh habitat, we identified four breeding populations (Figure 4-1). Three of those populations occur in South Carolina: North Inlet-Bulls Island,
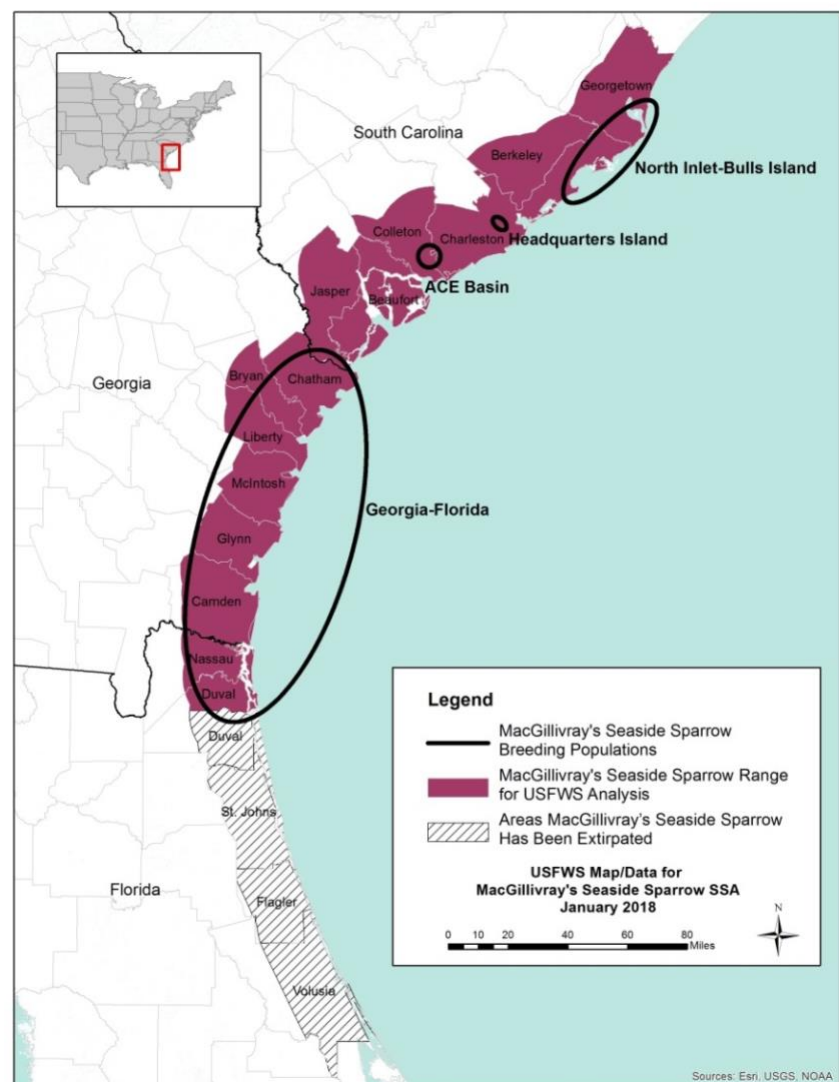
Figure 4-1. MacGillivray's seaside sparrow (*Ammodramus maritimus macgillivraii*) range map highlighting counties that are currently occupied in purple and areas presumed extirpated in black hatching. The four breeding populations of MacGillivray's seaside sparrow are labelled.

Headquarters Island, and ACE Basin. MacGillivray's seaside sparrows are found breeding in higher densities in the marshes that support these three populations relative to other marshes in South Carolina where individuals may occur in substantially lower densities or not at all. The fourth population, Georgia-Florida, is a relatively contiguous single population. Each breeding population is discussed below in north to south order.

4.1.1 North Inlet-Bulls Island

The North Inlet-Bulls Island population is the largest in South Carolina and occurs over a distance of approximately 40 miles (64 km), from Hobcaw Barony and the North Inlet-Winyah Bay National Estuarine Research Reserve in Georgetown, Georgetown County, to Bulls Island of the Cape Romain NWR in Awendaw, Charleston County; Figure 4-2. Marsh habitat is expansive in this area with large swaths of smooth cordgrass-dominated low marsh; however, individuals breed in the smaller, lower elevation areas of high marsh that include large patches of black needlerush. Breeding birds also are found in a few managed impoundments which are often brackish and non-tidal at the Tom Yawkey Wildlife Center Heritage Preserve (Yawkey) and Santee Coastal Reserve Wildlife Management Area (WMA; Santee Coastal) (Laskaris, 2016, pp. 12-14). The North Inlet-Bulls Island population is therefore patchily distributed and most individuals are primarily concentrated in grouped territories at a few known sites at Yawkey, Santee Coastal, and Cape Romain NWR (Bulls Island). There is no current population size estimate for this population or for the other two populations in South Carolina, nor at the state-level (see Appendix B for information regarding South Carolina abundance estimates).

The three properties (Yawkey, Santee Coastal, and Cape Romain NWR) contain historically occupied marshes, and were recently surveyed for MacGillivray's seaside sparrow in 2014-2015 during the breeding season (Hill & Laskaris, 2015, p. 1). Survey results found high occupancy for MacGillivray's seaside sparrows at Yawkey with 62% of survey points occupied in 2014 and 100% of survey points occupied in 2015, and more moderate occupancy at Santee Coastal (43% of points occupied in 2014 and 57% of points occupied in 2015; Hill & Laskaris, 2015, p. 2). No seaside sparrows were detected at the northern end of Cape Romain NWR in 2014 (no points were surveyed in 2015; Hill & Laskaris, 2015, p. 2); however, the surveyed points were in low marsh and the subspecies is not typically found in this marsh vegetation zone during the breeding season in South Carolina. Bulls Island at the southern end of Cape Romain NWR was surveyed in 2015 and 2017 during the breeding season, and a few MacGillivray's seaside sparrows were detected in both years (C. Hill, 2015, unpublished data; A. Smith & W. Wiest, 2017, unpublished data). North Inlet, a prime wintering site for *A.m. maritimus* and MacGillivray's seaside sparrow (more than 50+ individuals), also was surveyed during the breeding season, but only one individual of MacGillivray's seaside sparrow was detected (C. Hill, 2015, unpublished data). This result combined with the results from other wintering sites surveyed elsewhere in the state during the breeding season, provided further evidence that prime wintering sites do not serve as prime breeding sites and vice versa for South Carolina birds (Hill & Laskaris, 2015, p. 5; Sprunt, 1926, p. 549).

Figure 4-2. The approximate location of the North Inlet-Bulls Island breeding population of MacGillivray's seaside sparrow (*Ammodramus maritimus macgillivraii*) in South Carolina.

Nearly all marsh from North Inlet to Bulls Island is protected and managed as part of four conservation areas, that involve private (Hobcaw Barony owned by the Belle W. Baruch Foundation), State (Yawkey and Santee Coastal), and Federal (Cape Romain NWR) oversight (U.S. Geological Survey, Gap Analysis Program [USGS GAP], 2016, unpaginated; Table 3-1). Access or entry to some of the properties is only by guided tours or programs, and many areas are only accessible by boat. Collectively, over 133,000 acres (ac) (53,823 hectares [ha]) of large natural wetland complexes and smaller areas of uplands are protected, as well as managed impoundments that were former rice fields (NOAA, no date, p. 1; SCDNR, 2016a, unpaginated; SCDNR, 2016b, unpaginated; USFWS, 2016b, unpaginated).

4.1.2 Headquarters Island

The Headquarters Island population occurs on Headquarters Island, which is situated along Pennys Creek and the Stono River between James and Johns Islands in Charleston County, SC (Figure 4-3). The site is roughly equidistant, 25 miles (40 km) from Bulls Island to the northeast and Jehossee Island (Ashepoo, Combahee, and Edisto [ACE] Basin population) to the southwest, while it is located 8 miles (13 km) from the Atlantic Ocean. This population of MacGillivray's

seaside sparrow occupies an area of irregularly flooded, natural high marsh that is dominated by black needlerush (40% cover) and medium-height smooth cordgrass (30%; Hill & Post, 2005, p. 120). The remaining cover is saltgrass (10%); a mixture of glasswort, sea lavender (*Limonium carolinianum*), and short-form smooth cordgrass (15%); and multiple eroded dikes covered by shrubs (5%; Hill & Post, 2005, p. 120).

The population was first studied in 1998, and by 2001, 187 adults had been uniquely banded on the 24.7 ac (10 ha) study site; the authors reasoned that the captures constituted at least 95% of the breeding birds in the population (Hill & Post, 2005, p. 120). The activity spaces of the males were mapped almost daily from mid-April to mid-June during this period (Hill & Post, 2005, p. 120). Males defended small, overlapping nesting territories that ranged from 0.07-0.30 ac (mean = 0.14 ac, *n* = 40), and males and females foraged alone away from the nesting territory (Hill & Post, 2005, p. 124). Seaside sparrows exhibit a wide variation in nesting density and territorial defense (Post et al. 1983, p. 134) and the observations for this study population concluded that Headquarters Island birds occupy grouped territories (Hill & Post, 2005, p. 124). Individuals also exhibited high site fidelity and limited dispersal (less than 270 m on average; Post & Greenlaw, 2009, unpaginated). This population was briefly revisited during the 2015 breeding season during which time birds were found nesting in dense concentrations (Hill & Laskaris 2015, p. 3).
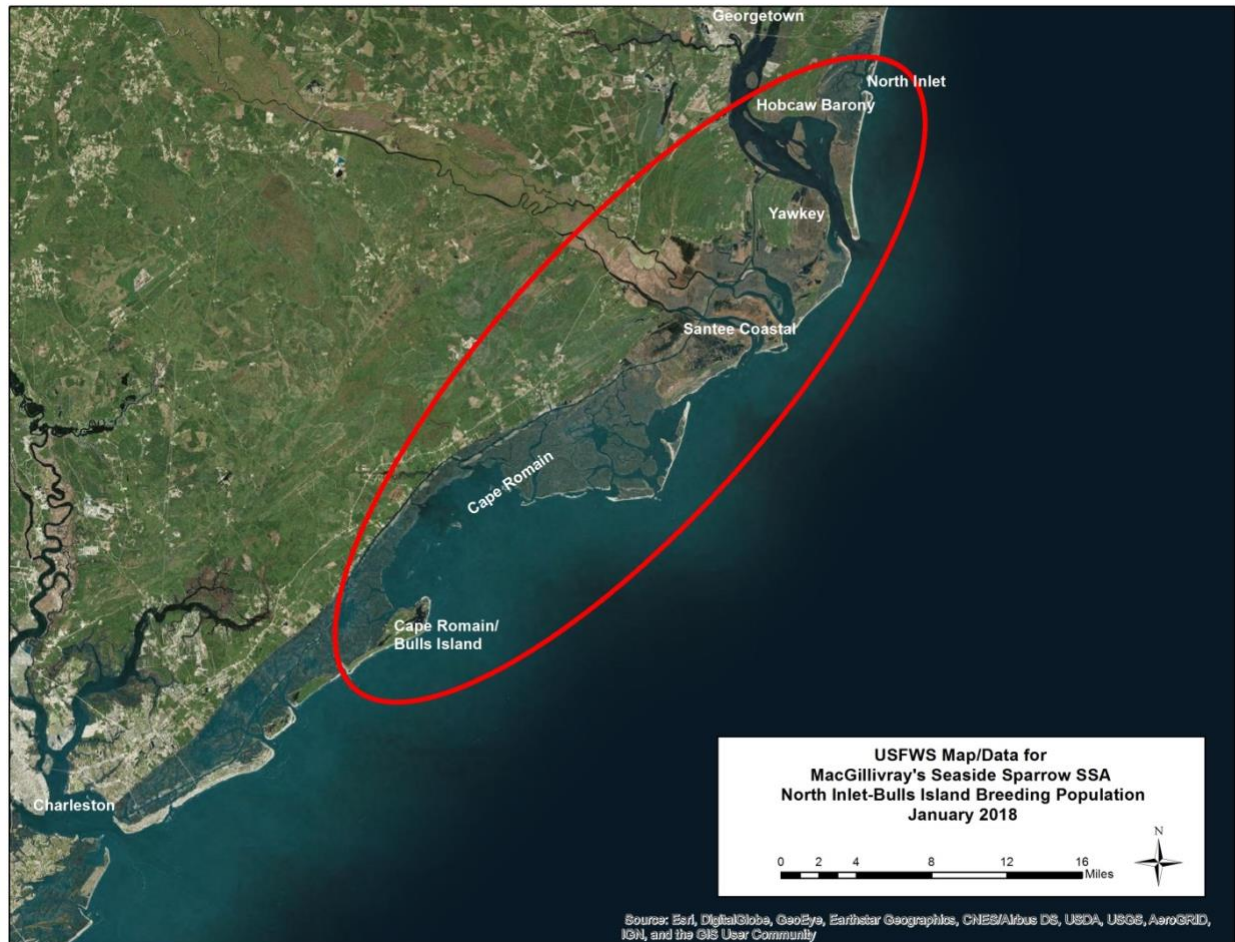
Figure 4-3. The approximate location of the Headquarters Island breeding population of MacGillivray's seaside sparrow (*Ammodramus maritimus macgillivraii*) in South Carolina.

It is hypothesized that the Headquarters Island population is the "descendent" of the Rantowles colony, first described in 1924 (W. Post, pers. comm. in Hill & Laskaris, 2015, p. 3; Sprunt, 1924, entire). The Rantowles colony was the first breeding colony of MacGillivray's seaside sparrow discovered in South Carolina, approximately 15 miles southwest of the City of Charleston near Rantowles (Sprunt, 1924, p. 482). The currently occupied Headquarters Island site is approximately 7.6 miles (12.2 km) downriver from the Rantowles site (Hill & Laskaris, 2015, p. 3).

Headquarters Island is influenced by rapid human population growth and industrial development in the Charleston Metropolitan Area. The site is only 3-4 miles from downtown Charleston. The majority of upland forest on Headquarters Island has been converted into residential development and development is ongoing. Land use/land cover in the Stono River Watershed (03050202-02) is approximately 29% forested land, 23% nonforested wetland, 16% forested wetland, 13% urban land, 10% water, 8% agricultural land, and 1% barren land (South Carolina Department of Health and Environmental Control [SCDHEC], 2013, p. 119), with many development projects underway. Tidal marshes on Headquarters Island are not in a formal protected area or conservation easement (USGS GAP, 2016, unpaginated), but are protected through existing regulatory protections for wetlands (e.g., Clean Water Act, South Carolina Coastal Tidelands and Wetlands Act).

4.1.3 ACE Basin

The ACE Basin refers to the confluence of the Ashepoo, Combahee, and Edisto (ACE) Rivers in southeastern South Carolina, which form the Saint Helena Sound. Within the ACE Basin, MacGillivray's seaside sparrows are found in natural high marsh and in impoundments (managed wetlands that were former rice fields) at Bear Island WMA and Jehossee Island (Jehossee): Figure 4-4. Jehossee has historically supported breeding MacGillivray's seaside sparrows in black needlerush and clumps of sand cordgrass (Hill, 2014, p. 7). The subspecies is present in similar areas at Bear Island (C. Hill, 2014, unpublished data), which is located immediately due west of Jehossee across the Edisto River; the River forms the county boundary between Bear Island in Colleton County and Jehossee in Charleston County. MacGillivray's seaside sparrows were detected at Bear Island during breeding season surveys in 2014 (C. Hill, 2014, unpublished data). No MacGillivray's seaside sparrows were recorded on Jehossee during 2014 surveys; however, the habitat had recently been altered due to a burn and a few individuals were observed outside the formal survey period (Hill & Laskaris, 2015, p. 3). Jehossee is presumed to support a high nesting density of MacGillivray's seaside sparrow in most years (M. Purcell, L. Hartis pers. comm. in Hill & Laskaris, 2015, p. 3). ACE Basin NWR was revisited during the 2017 breeding season and MacGillivray's seaside sparrows were detected at 21% of survey points (11 of 52 points) with an average of 2.0 birds detected at each occupied point (A. Smith & W. Wiest, 2017, unpublished data).

Bear Island is managed by South Carolina Department of Natural Resources and Jehossee Island is managed as part of the ACE Basin NWR by the Service (Table 3-1). These properties along with others in the ACE Basin form one of the largest undeveloped complexes of wetlands and upland habitat on the east coast of the United States (SCDNR, 2011, p. 67). For this reason, the ACE Basin National Estuarine Research Reserve was established in 1992 and includes Bear Island and Jehossee in the approximately 94,621-ac (38,292-ha) large Reserve boundary (SCDNR, 2011, pp. 4-5). In 2011, 75 landowners owned 75% of the land in the ACE Basin that was east of Highway 17 (SCDNR, 2011, p. 62). However, the region is facing increasing development pressure from the rapidly growing Charleston to the north and Beaufort to the south (SCDNR, 2011, pp. 4, 62).

Figure 4-4. The approximate location of the ACE Basin breeding population of MacGillivray's seaside sparrow (*Ammodramus maritimus macgillivraii*) in South Carolina.

4.1.4 Georgia-Florida

Coastal marshes in Georgia and northeast Florida support one contiguous population of MacGillivray's seaside sparrow; Figure 4-5. This population is being analyzed as two separate analysis units for this SSA: 1) a Georgia Analysis Unit and 2) a Florida Analysis Unit. Within Georgia, the analysis unit comprises the coastal counties of Chatham, Bryan, Liberty, McIntosh, Glynn, and Camden counties. Georgia's coastline is approximately 100 miles (160 km) in length from the Savannah River to the St. Marys River and contains 368,484 ac (149,120.184 ha) of estuarine wetlands (GADNR, 2012, p. 31). Within Florida, the analysis unit is limited to the counties of Nassau and Duval, and the current occupied range extends approximately 20 miles (32 km) from the St. Marys River to the St. Johns River (Schwarzer & Cox, 2017, p. 10). The St. Marys River, the boundary between Georgia and Florida, is not considered a barrier to MacGillivray's seaside sparrow movement, and therefore, individual birds likely move among marshes between the two states (A. Schwarzer, 2017, pers. comm.).
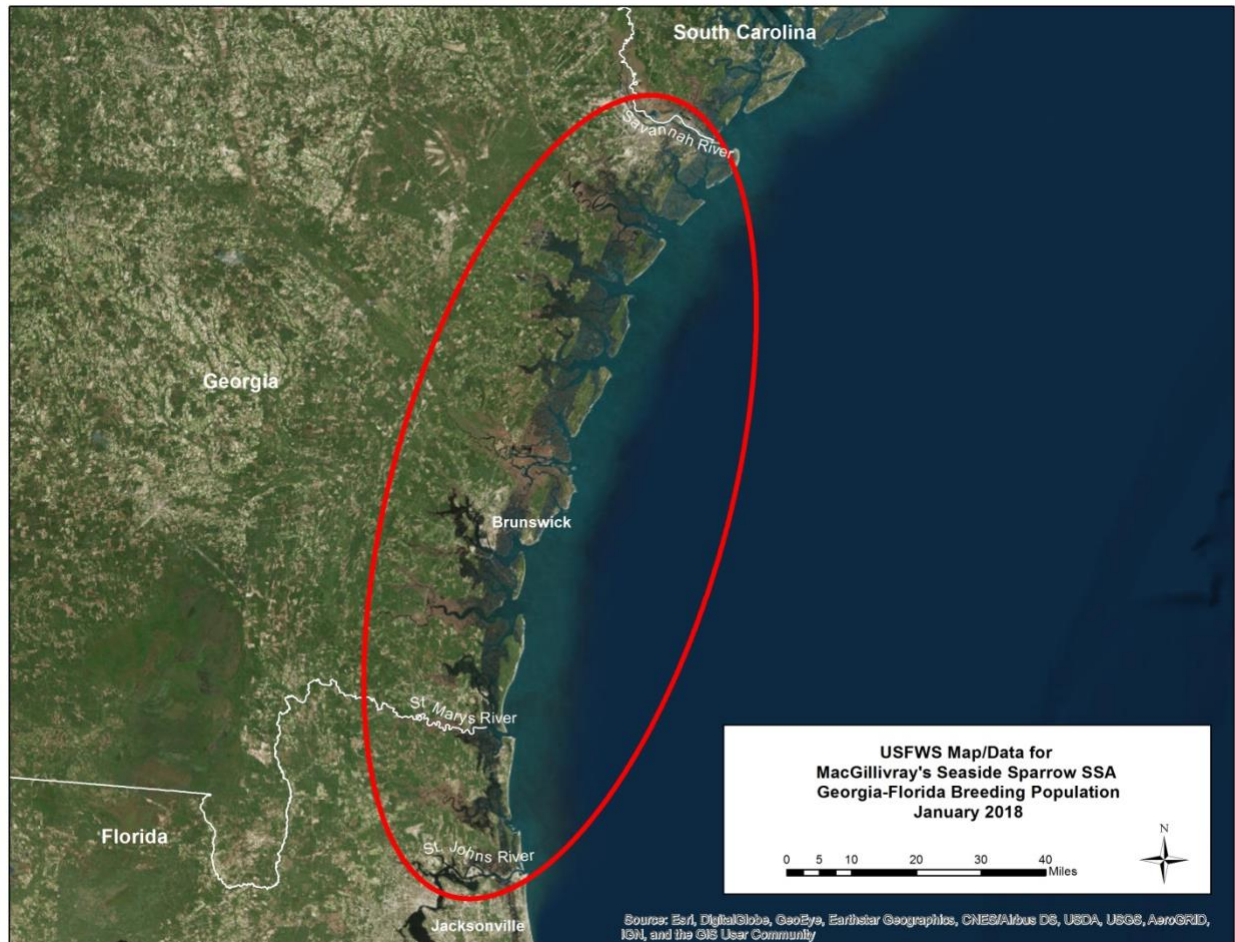
Figure 4-5. The approximate location of the Georgia-Florida breeding population of MacGillivray's seaside sparrow (*Ammodramus maritimus macgillivraii*).

Unlike the South Carolina populations, MacGillivray's seaside sparrows in Georgia and Florida use salt marshes as habitat, and brackish marshes less so, and do not use impoundments (Hunter, et al., 2016a, pp. 3-5; T. Schneider, A. Schwarzer, 2017, pers. comm.). In Georgia, MacGillivray's seaside sparrow was found to be most abundant in higher elevation areas of low salt marshes that were far from forests and development (Hunter, et al., 2016a, p. 3; Hunter, et al., 2017, p. 24). In Florida, the subspecies uses similar habitats as Georgia birds, with a tendency to remain in higher elevations of low natural salt marsh far from trees/uplands, and in areas without succulent vegetation (Schwarzer & Cox, 2017, p. 2).

Historical population estimates are only available for the Florida analysis unit of the Georgia-Florida population of MacGillivray's seaside sparrow. Historical population size estimates were limited to an estimated 750 pairs for the Florida analysis unit in the 1970s and 1980s (McDonald 1988, p. 54). However, a current population estimate for the Florida analysis unit was not available, so a comparison with the historical estimate could not be made. A current population size estimate was only available for the Georgia analysis unit of the Georgia-Florida population. In Georgia, the current MacGillivray's seaside sparrow population size is estimated to be 32,800

birds (95% confidence interval: 25,200-40,400 birds; Hunter et al. 2016a, p. 3). The Georgia analysis unit is likely the stronghold of MacGillivray's seaside sparrow considering the restricted range of the Florida analysis unit and lower occupancy estimates in South Carolina (Hunter et al. 2016a, p. 6).

The tidal ecosystem is a distinguishable characteristic of the Georgia and northeast Florida coast. The Georgia coast is in the approximate center of the curved coastline known as the South Atlantic Bight which extends from Cape Lookout, NC to West Palm Beach, FL (Atkinson, et al., 1985 in Bacopoulos & Hagen, 2017, p. 13). Variation in bathymetry along the-Bight causes subsequent variation in tides, and the inshore features of the coastline (i.e., tidal inlets, estuarine rivers, and tidal zones) enhance the significance of lunar semi-diurnal tidal circulation. Tides within the Bight are particularly enhanced from Winyah Bay, South Carolina to the Georgia/Florida border (Bacopoulos & Hagen, 2017, p. 13). As the tide approaches the Atlantic coast it reaches the northern portion of the Bight first. As the water makes its way towards the center of the Bight along the Georgia coast, the water follows the bathymetry and increases in elevation. Coastal communities at the fringes of the Bight, such as Cape Hatteras, NC and Miami, FL, generally have two-foot tides, but by the time the water reaches the Georgia coast the high and low tides rise and fall between approximately six to nine feet twice a day. The tidal amplitude in northeast Florida typically ranges between four and six feet.

The Georgia coastline consists of 14 primary barrier islands that shelter extensive back-barrier lagoon marsh systems from waves and currents. The islands from north to south are: Tybee, Little Tybee, Wassaw, Ossabaw, St. Catherines, Blackbeard, Sapelo, Wolf, Little St. Simons, Sea, St. Simons, Jekyll, Little Cumberland, and Cumberland (the largest of Georgia's barrier islands). Most of Georgia's barrier islands are protected by the state or federal government or conservation friendly private ownership (e.g., Little St. Simons and St. Catherines). Some islands have been conserved as NWRs and National Wildernesses, and one - Cumberland Island - is a National Seashore. Only four of these islands are accessible by car while the rest are only accessible by boat or floatplane (Jackson & Stakes, 2004, p. 374).

Northeast Florida (from the Florida/Georgia border to the St. Johns River) contains 50,860 ac (20,582 ha) of salt and brackish marsh (A. Schwarzer, 2017, pers. comm.). Of this, 24,433 ac (9,887 ha; 48%) are within conservation areas including federal, state and municipal lands and private non-profits such as The Nature Conservancy. The largest entity is the Timucuan Ecological and Historic Preserve, a conglomerate of public and private lands managed by the National Park Service. The remaining 26,427 ac (10,694 ha; 52%) are in unprotected, private land ownership.

4.1.5 Areas Presumed Extirpated

MacGillivray's seaside sparrow (including the synonymous Smyrna seaside sparrow) once occupied approximately 370 miles (595 km) of coastline in the southeastern U.S. and is believed to be extirpated from approximately 27% (100 miles or 160 km) of its historic range. Areas from

which the subspecies has been extirpated are all in Florida, including Duval County south of the City of Jacksonville, and St. Johns, Flagler, and Volusia Counties, Florida (Kale, 1983, pp. 42-45; McDonald, 1988, p. 55) (Figure 4-6). The marshes in these areas supported either a single isolated population (Rising, 2005, p. 493) or multiple, small scattered populations (McDonald, 1988, pp. 122-128).

The range contraction was first documented in Volusia County with the loss of suitable habitat from invasion by mangroves (Nicholson 1946 & 1950 in Kale, 1983, p. 44). In 1939, hundreds of birds still bred in the marshes at New Smyrna Beach but no breeding birds were detected there from 1948-1949 (Nicholson 1950 in Kale, 1983, p. 44). Heavy spraying of DDT in the late 1940s and early 1950s also may have contributed to the decline (Austin, 1968, p. 836), but evidence is lacking and the birds were thought to have already been gone by the time DDT was used extensively in these marshes (Kale, 1983, p. 44). By 1959, no birds were found nesting south of Matanzas Inlet in St. Johns County (Austin, 1968, p. 836), and the subspecies was presumed extirpated south of the St. Johns River in Duval County by 1976 (Kale, 1983, pp. 44-45).

Because of the continued presence of suitable habitat between Matanzas Inlet and St. Augustine (St. Johns County), subsequent breeding season surveys were performed there up to year 2017 (Kale, 1996, p. 610; NeSmith & Jue, 2003, p. 1; Schwarzer & Cox, 2017, entire). Resurveys from 1979-1981 and 1987-1988 found no birds breeding south of the City of Jacksonville (i.e., the St. Johns River; McDonald, 1988, pp. 54-55). Surveys in 2000-2001 yielded a maximum count of three birds at a site in northern Flagler County; this site was 60 miles south of the nearest known breeding location in Duval County (NeSmith & Jue, 2003, p. 24). One additional bird was detected in southern Duval in 2000, but no birds were detected when the area was resurveyed in 2001 (NeSmith & Jue, 2003, p. 18). Finally, surveys performed in 2016-2017 revealed no additional sightings. Based on the survey results ranging from 1979-2017, experts have inferred that MacGillivray's seaside sparrow is functionally extirpated from the southern portion of its historic range (i.e., south of the St. Johns River in Duval County to Volusia County, Florida; Schwarzer & Cox, 2017, p. 2).
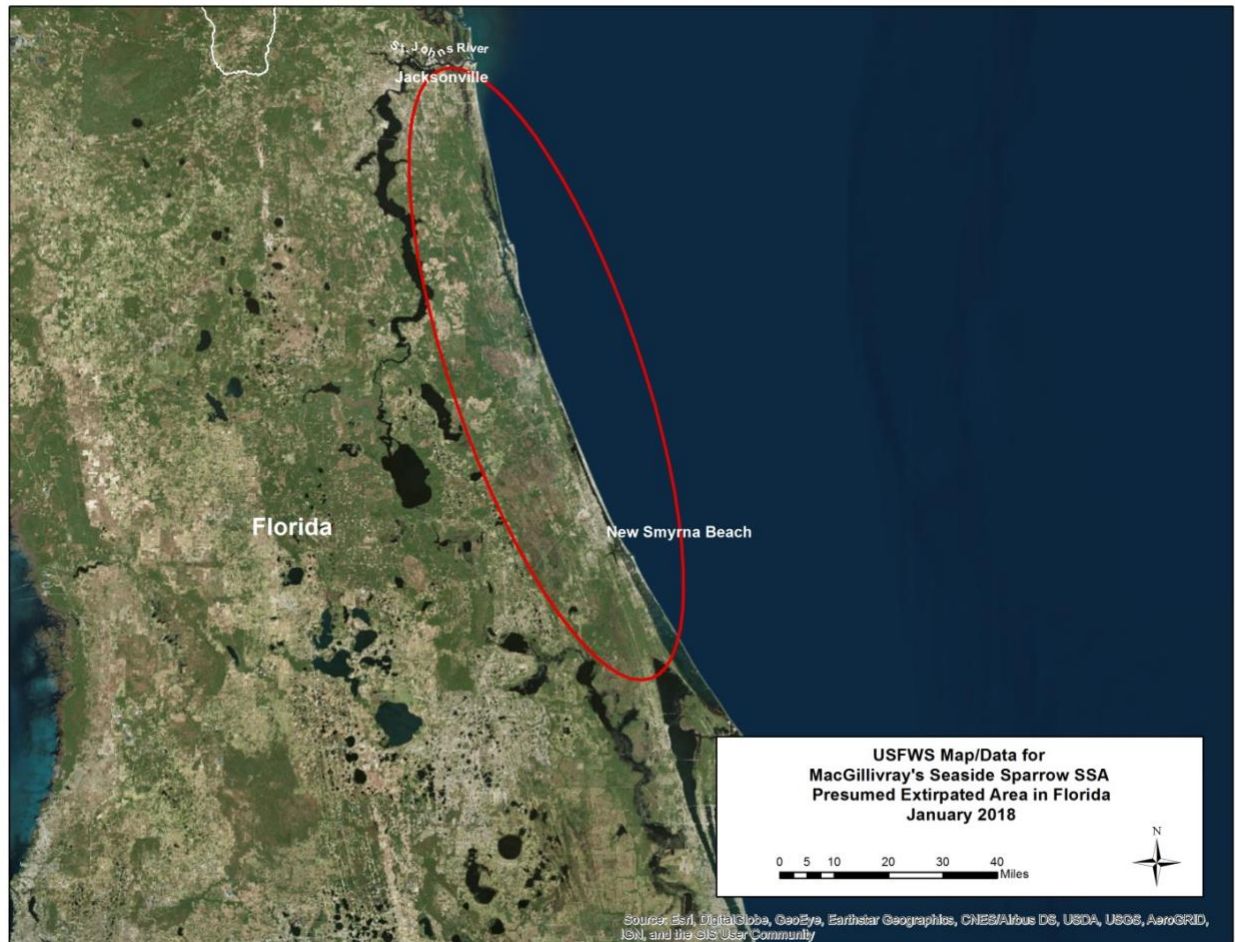
Figure 4-6. The approximate location of the area where MacGillivray's seaside sparrow (*Ammodramus maritimus macgillivraii*) is presumed to be extirpated in Florida.

4.2 Methods for Estimating Current Condition

For the purpose of this assessment, we defined **viability** as the ability of MacGillivray's seaside sparrow to sustain populations in the wild beyond a biologically meaningful time frame. Using the SSA framework, we described viability of MacGillivray's seaside sparrow by estimating the current condition, and (later) predicting the future condition, of metrics used to assess **resiliency**, **representation, and redundancy**.

4.2.1 Population Resiliency

Given data availability, MacGillivray's seaside sparrow resiliency was estimated using population-level demographics (abundance, reproduction, and population growth). To assess current population resiliency, we estimated the maximum number of birds detected at a survey point (as a proxy for abundance), nest predation rates, nest inundation rates, daily/period nest survival rates, and population growth rates by analysis unit. Analysis units in South Carolina were equivalent to population units for abundance and reproduction estimates, but a single analysis unit at the State-scale was analyzed for population growth. Because data in Georgia and

Florida were collected separately, the Georgia-Florida population was split into three separate analysis units: Georgia, the non-extirpated range in Florida, and the extirpated range in Florida. Abundance and reproductive values were estimated from existing survey and demographic data from across the subspecies' range (Hill & Laskaris, 2015, p. 1-2; Hunter, et al., 2016b, pp. 137-138; Hunter, et al., 2017, p. 21; Laskaris, 2016, p. 23; Schwarzer & Cox, 2017, p. 5; E. Hunter, A. Schwarzer, A. Smith, W. Wiest, 2017, unpublished data).

To assess current condition of the populations, we analyzed survival rates for adults, juveniles, and nests, to estimate population growth. We calculated an upper, mid, and lower range estimate of population growth (λ [lambda]) for each analysis unit using the greatest (upper), mean (mid), and lowest (lower) estimates of the survival and reproductive terms used to calculate population growth. Population growth equaled the sum of apparent annual adult and first year survival probabilities multiplied by the product of fecundity and the proportion of the mean number of fledglings per successful nest that were female. We calculated population growth rates assuming a finite rate of increase/decrease for each population (analysis unit) in one time step (= 1 year). Population growth rates indicated whether a population was currently declining ($0 < \lambda < 1.0$), stable ($\lambda = 1.0$), or increasing ($\lambda > 1.0$). We considered the mid-range estimates of population growth to be the most realistic estimate (i.e., the weighted average among years) for each MacGillivray's seaside sparrow analysis unit.

Our apparent annual adult survival rates were based on published survival rates for the *A. m. maritimus* subspecies in New Jersey (61.6%, 95% CI: 52.5-70.0%; Roberts, et al., 2017, p. 122) and in Connecticut (72.0-89.6%; Borowske, 2015, p. 104). The majority of adult mortality for passerines is believed to occur during migration (e.g., Gill, 2007, p. 512; Sillett & Holmes, 2002, p. 304), and because *A. m. maritimus* migrates while *A. m. macgillivraii* is considered non-migratory, we assumed MacGillivray's seaside sparrow has, on average, higher apparent annual adult survival than northern seaside sparrow populations. Therefore, we considered the greatest adult survival estimate from Connecticut (89.6%) to be our upper-range value, the upper bound of the 95% CI from New Jersey (70%) to be our mid-range value (which was similar to the lower bound for Connecticut [72%]), and the mean estimate from New Jersey (61.6%) to be our lower-range value.

To estimate apparent annual first year (juvenile) survival we used empirical data for our mid and lower-range values. Our mid-range value for first year survival probability (0.35) was based on a published juvenile survival estimate for seaside sparrows in New York (Post & Greenlaw, 1982, p. 105). Our lower-range first year survival probability estimate was based on juvenile return rates for marsh wrens that nest sympatrically with MacGillivray's seaside sparrows in the Florida analysis unit; post-fledging 21-day survival for marsh wrens was approximately 6% (A. Schwarzer, 2017, unpublished data). We increased the 6% juvenile marsh wren survival rate to 10% (= 0.10 probability) for seaside sparrows for our lower-range estimate since juvenile survival in songbirds increases with adult body mass and nestling period duration (Cox, et al., 2014, p. 187), and seaside sparrows are larger than marsh wrens. It is best to use empirically derived survival rates, as we have done for our mid and lower-range estimates, to minimize inaccurate estimates of annual population growth (Cox, et al., 2014, p. 183). However, many studies of passerines lack information on juvenile survival and consequently, estimate annual juvenile survival as 25-50% the rate of annual adult survival (Cox, et al., 2014, p. 183; Donovan

& Thompson III, 2001, p. 874). To this end, we estimated our upper-range juvenile survival rate as 50% (= 0.45) of our upper-range adult survival rate. We held survival constant across MacGillivray's seaside sparrow analysis units within each range (upper, mid, lower).

We calculated fecundity by multiplying period nest survival probability, mean number of nest attempts per breeding pair, and mean number of fledglings per successful nest. Upper and lower range nest survival rates for Georgia and Florida were estimated based on observed nesting data, while mid-range nest survival rates represent the average of nest rates based on the observed data. For example, period nest survival was estimated for each of three years in Georgia, and it was *high* in one year (17.5%) but *low* in the other two years (1.5% and 2%; period nest survival calculated from daily nest survival rates in Hunter, et al., 2016b, p. 138). Thus, mid-range period nest survival was calculated using the weighted average of high and low end parameters (2*low + 1*high = mid). In South Carolina, only one year of nesting data was available, therefore the mid-range period nest survival represents the observed value from a single breeding season and the upper and lower nest survival rates represent the observed value ±1 standard error (Laskaris, 2016, p. 23). For mean number of nest attempts per breeding pair, we kept this value constant (5 nest attempts) based on observations of MacGillivray's seaside sparrow in the Georgia analysis unit (Hunter, 2017, pp. 462-463). For mean number of fledglings per successful nest, we used the same observed nesting data that was used to calculate nest survival; we held the term constant across the three ranges for each analysis unit (South Carolina = 2.50 mean number of fledglings per successful nest, Georgia = 1.90, Florida = 2.47). We then multiplied fecundity by the proportion of the mean number of fledglings per successful nest that were female for the reproductive term in the population growth rate equation. We assumed an equal sex ratio (0.5) across ranges and analysis units. The reproductive term was then multiplied by the sum of apparent annual adult and first year survival probabilities to estimate population growth, as described above.

To further assess potential population growth rates for the South Carolina, Georgia, and Florida analysis units, the Florida Fish and Wildlife Conservation Commission (FWC) conducted an additional analysis to simulate population growth rates (λ) for each analysis unit (Tornwall & Schwarzer, 2018; Appendix C). This analysis examined the proportion of population growth rates calculated that were less than 1.0 out of 100,000 trials per analysis unit (Tornwall & Schwarzer, 2018; Appendix C). Each trial selected randomly selected values for adult survival, juvenile survival, and nest survival using the ranges described above.

4.2.2 Subspecies Representation and Redundancy

Representation reflects a species' adaptive capacity such that measures of genetic and ecological variability capture this metric. For MacGillivray's seaside sparrow, we used three metrics to estimate and predict representative units that reflect the subspecies' adaptive capacity: 1) genetic variability, 2) habitat variability, and 3) tidal variability.

MacGillivray's seaside sparrow exhibits adaptive potential by maintaining genetic diversity between populations and utilizing three distinct habitat types during the breeding season, smaller patches of lower elevation areas of high marsh (in South Carolina), impoundments (in South Carolina), and higher elevation areas of low marsh (in Georgia and Florida; C. Hill, E. Hunter, T.

Schneider, and A. Schwarzer, 2017, pers. comm.). Additionally, we used the metric of tidal variability to reflect MacGillivray's seaside sparrow use of habitat with varied tidal regimes caused by the shape of the South Atlantic Bight. Coastal areas that MacGillivray's seaside sparrows occupy within the South Atlantic Bight can experience on average between four to nine-foot tidal regimes (higher in Georgia and Florida, lower in South Carolina), with a maximum of approximately six and a half to ten feet in the middle of the Georgia coast (Wiegart & Freeman, 1990, p. 4). Therefore, representative elements for MacGillivray's seaside sparrow include three genetically distinct units, three habitat units, and tidal variability throughout the subspecies' range. To maintain existing adaptive capacity (representation), the subspecies should have resilient populations across the representative units.

The metric of redundancy reflects a subspecies' ability to remain extant after experiencing extreme catastrophic events. Redundancy is measured by assessing the number and distribution of resilient populations throughout a subspecies' range. Species (and subspecies) that are well-distributed across their historical range are considered less susceptible to extinction and more likely to be viable than species confined to a small portion of their range (Carroll, et al., 2010, entire; Redford, et al., 2011, entire). We evaluated the current distribution of MacGillivray's seaside sparrow populations through their present-day spatial locations. To have high redundancy, MacGillivray's seaside sparrow would need to have resilient populations spread throughout its range.

4.3 Current Condition Results

4.3.1 Current Population Resiliency

To assess the current condition of MacGillivray's seaside sparrow, we estimated abundance, reproduction, and population growth rates by analysis unit (Tables 4-1, 4-2, and 4-3). In South Carolina populations surveyed from 2014-2015, the mean occupancy at sampled sites ranged from 14 to 44% with approximately 1 to 2 birds at each occupied site (Table 4-1). Additional breeding season surveys in 2017 yielded 1 bird per occupied point at Cape Romain NWR (10% of survey points; North Inlet-Bulls Island population) and 2.0 birds per occupied point at ACE Basin NWR (21% of survey points; ACE Basin population) (Table 4-2). Occupancy in the Georgia analysis unit was slightly higher (51%) than in the Florida analysis unit (41%) with an average of 3.3 to 2.5 birds at each occupied point, respectively (Table 4-1). It should be noted that the abundance estimates for the South Carolina populations and Georgia analysis unit (and the NWRs) were from survey data using a 200-m point count radius while the Florida analysis unit estimates were from 100-m point count radius surveys (Hill & Laskaris, 2015, pp. 1-2; Hunter, et al., 2017, p. 21; Schwarzer & Cox, 2017, p. 5; A. Smith & W. Wiest, 2017, unpublished data).

Table 4-1. MacGillivray's seaside sparrow (*Ammodramus maritimus macgillivraii*) abundance during the breeding season by analysis unit. Abundance is reported as the average number of the maximum number of birds detected at each survey point. Values represent the current condition of each MacGillivray's seaside sparrow analysis unit based on data collected from 2013-2016. p/o = portion of.

| State | Abundance* (Number of Survey Points) | |
Analysis Unit	All Points	Occupied Points
South Carolina	0.5 (129)	2.0 (30)
North Inlet-Bulls Island	0.9 (62)	2.1 (27)
Headquarters Island	No data	
ACE Basin	0.1(14)	1.0 (2)
Georgia		
Georgia coastline	1.3 (214)	3.3 (110)
Florida		
Nassau and p/o Duval counties	1.2 (62)	2.5 (26)
St. Johns, Flagler, Volusia, and p/o Duval counties	Ø	Ø

*Abundance was estimated using a 200-m point-count radius in South Carolina and Georgia and a 100-m radius in Florida. Survey data from: Hill & Laskaris, 2015, pp. 1-2; Hunter, et al., 2017, p. 21; Schwarzer & Cox, 2017, p. 5.

Table 4-2. MacGillivray's seaside sparrow (*Ammodramus maritimus macgillivraii*) abundance during the breeding season at National Wildlife Refuges. Abundance is reported as the average number of the maximum number of birds detected at each survey point. Values represent the current abundance of MacGillivray's seaside sparrow at each refuge based on data collected in 2017; refuges are listed north to south following the coastline. Corresponding MacGillivray's seaside sparrow breeding populations are listed for reference; n/a = not applicable. Source: A. Smith & W. Wiest, 2017, unpublished data.

| State | National Wildlife Refuge* | Abundance[†] (Number of Survey Points) | | MacGillivray's Seaside Sparrow Breeding Population |
		All Points	Occupied Points	
South Carolina	Cape Romain	0.1 (31)	1 (3)	North Inlet-Bulls Island
South Carolina	ACE Basin	0.4 (52)	2.0 (11)	ACE Basin
South Carolina	Pinckney Island	0 (16)	-	n/a
South Carolina / Georgia	Savannah	0 (31)	-	n/a
Georgia	Wassaw	0.6 (20)	1.9 (7)	Georgia-Florida
Georgia	Harris Neck	0 (16)	-	Georgia-Florida

*The subspecies may not be present at Savannah National Wildlife Refuge during the breeding season. Tybee, Blackbeard Island, and Wolf Island National Wildlife Refuges support the subspecies, but were not surveyed.
[†]Abundance was estimated using a 200-m point-count radius.

Daily nest survival and consequently period nest survival was highest (93.1% and 18.0%, respectively) in the North Inlet-Bulls Island population of South Carolina; however, the rates are based on a small sample size from a single high abundance site (Tom Yawkey Wildlife Center) during one breeding season (Table 4-3). There is no recent survey or demographic data for Headquarters Island, but anecdotal observations during the 2015 breeding season reported birds nesting in dense concentrations similar to birds nesting at Yawkey (Hill & Laskaris, 2015, p. 3). Nest survival in the Georgia-Florida population (daily nest survival = 86.7 and 86.5% and period nest survival = 3.3 and 3.1% for the Georgia and Florida analysis units, respectively) was lower than in South Carolina (daily nest survival = 93.1 and period nest survival = 18) (Table 4-3). Nest predation and inundation was highest in Georgia > Florida > South Carolina and ranged from 63% to 28% for predation and 17% to 0% for inundation, respectively. Overall, nest survival for MacGillivray's seaside sparrow is lower than for the subspecies' northern counterpart, *A. m. maritimus*, where daily nest survival rates range annually from 88 to 97% in Maryland (Kern, et al., 2012, p. 937), 92 to 97% in New Jersey (Roberts, et al., 2017, p. 124), and 95% in Connecticut (Gjerdrum, et al., 2005, p. 857).

Based on demographic work completed between 2013 and 2017, the Georgia-Florida population shows evidence of low reproduction and period nest survival (Table 4-3), and, in turn, most of the population growth rates indicate that the Georgia-Florida population is declining (i.e., population growth rate < 1.0) (Table 4-4). The South Carolina analysis unit (North Inlet-Bulls

Island) shows evidence of maintaining a stable (1.09) and slightly increasing (1.43) population growth rate under the mid- and upper-range estimates, respectively; under the lower-range estimate, the South Carolina analysis unit shows a decline in population growth (Table 4-4). However, inter-year variation in nest survival can be significantly different and population growth can fluctuate above and below the 1.0 threshold among years. This is evidenced in the period nest survival in Georgia, where overall period nest survival was 3.3%, but the mean annual period nest survival ranged from 1.5% (95% CI: 0.6–3.5%) in 2015 to 17.5% (95% CI: 8.0–29.2%) in 2014 (calculated using a 24-day nest period and the daily nest survival rates reported in Hunter, et al., 2016b, p. 138). The mid-range population growth rate was calculated using the weighted average of high and low end parameters from three consecutive study years in the Georgia analysis unit, so it is realistic to consider a downward trend but not necessarily a constant population growth rate of 0.73.

The additional FWC analysis modeled population growth rates to develop a distribution of likely population growth values for the three analysis units (Tornwall & Schwarzer, 2018; Appendix C). Results determined that 34% of the simulated population growth rate values for South Carolina were < 1.0 while 66% of values were ≥ 1.0, indicating that this analysis unit is likely stable or increasing. For Georgia and Florida, 85% and 79% of trials estimated a population growth rate < 1.0, respectively, indicating that these analysis units are likely undergoing population declines under current conditions.

Overall, MacGillivray's seaside sparrow populations in South Carolina are likely stable and have higher resiliency than the Georgia-Florida population. Abundance was higher in the Georgia-Florida population (2.5-3.3 mean number of birds per occupied point) compared to the South Carolina populations (1-2 birds per occupied point; Table 4-1), but nest survival was lower in Georgia and Florida than in South Carolina (South Carolina data was based on a single breeding season compared to three breeding seasons for Georgia and Florida). Nest survival is a known reproductive constraint for passerines and low nest survival is a leading driver of population declines (e.g., Donovan & Thompson III, 2001, entire). A population viability analysis (PVA) for seaside sparrows in New Jersey found that the population went quasi-extinct in 37 years or less when fecundity was ≤ 1.24, adult survival = 0.60, and juvenile survival = 0.30 (the PVA was simulated under a 0.35 m rise in sea-level; Roberts, 2016, p. 67-68). Two of the three fecundity estimates (mid-range) for MacGillivray's seaside sparrow populations are currently less than 1.24, and this is under present day sea-level conditions (Table 4-4). Furthermore, two of the three estimates of population growth (mid-range) for MacGillivray's seaside sparrow are below 1.0, and this is assuming relatively high annual adult and juvenile survival (adult = 0.70, juvenile = 0.35), thus indicating that low nest survival is driving the downward population trend for MacGillivray's seaside sparrow in the Georgia and Florida analysis units. This is further supported by the additional FWC modeling which found the Georgia and Florida analysis units are currently undergoing population declines (Appendix C). For the upper-range estimates of population growth (which are unlikely due to very high survival rates for both adults and juveniles), all analysis units had estimates at or above 1.0 (Table 4.3); conversely, all analysis units experienced declines in population growth under the lower-range estimates.

When considering the ranges of the estimated population growth rates, the Georgia-Florida population is experiencing negative population growth, leading to an overall decline in the

current resiliency of this MacGillivray's seaside sparrow population and below the threshold needed to maintain population stability (i.e., $\lambda = 1.0$). Based on higher nest survival rates and stable population growth rates, the North Inlet-Bulls Island population in South Carolina has higher resiliency than the Georgia-Florida population. Current nesting data was unavailable for the two other populations in South Carolina (i.e., Headquarters Island and ACE Basin) so we were unable to estimate population growth rates; however, survey data and anecdotal observations show that MacGillivray's seaside sparrows in these populations are present at similar abundances and nest in dense concentrations as are found in the North Inlet-Bulls Island population. Therefore, we infer that the Headquarters Island and ACE Basin populations also have higher resiliency than the Georgia-Florida population and are likely stable in their current state.

Table 4-3. Reproduction rates for MacGillivray's seaside sparrow (*Ammodramus maritimus macgillivraii*) analysis units. Values represent the current condition of each analysis unit based on data collected from 2013-2017. p/o = portion of.

State Analysis Unit	Total Nests (*n*)	Successful Nests (*n*)	Nests Predated (*n*)	Nests Inundated (*n*)	Nests Abandoned (*n*)	Nest Failure – Cause Unknown (*n*)	Nest Fate Unknown (*n*)	Total Productivity (chicks fledged/total nests)	Daily Nest Survival[†] (%)	Period Nest Survival [‡] (%)
South Carolina										
North Inlet-Bulls Island[*]	36	16	10	0	0	8	2	1.1	93.1	18.0
Headquarters Island						No data				
ACE Basin						No data				
Georgia[*]										
Georgia coastline	354	51	224	61	4	10	4	0.3	86.7	3.3
Florida										
Nassau and p/o Duval counties	123	19	49	7	1	46	1	0.4	86.5	3.1
St. Johns, Flagler, Volusia, and p/o Duval counties	Ø	Ø	Ø	Ø	Ø	Ø	Ø	Ø	Ø	Ø

[*]Nest estimates for North Inlet-Bulls Island were from Tom Yawkey Wildlife Center, and for Georgia from a site with high abundance near Brunswick, GA.

[†]South Carolina and Georgia rates were calculated using MCestimate (Hunter, et al., 2016b, pp. 137-138; Laskaris, 2016, p. 23) and the Florida rate was calculated using logistic-exposure (A. Schwarzer, 2017, unpublished data).

[‡]Period nest survival was calculated using a 24-day nest period (3-day laying period, 11-day incubation period, and 10-day nestling period; Schwarzer & Cox, 2017, p. 9).

Table 4-4. Population growth rates for MacGillivray's seaside sparrow (*Ammodramus maritimus macgillivraii*) analysis units. Values represent the current condition based on data collected from 2013-2017. The Mid-Range estimates of population growth were calculated using the weighted average of high and low end nest survival rates when multi-year data was available.

Range[*]	Analysis Unit[†]	Apparent Annual Adult Survival Probability[‡]	Apparent Annual First Year Survival Probability[‡]	Period Nest Survival Probability[§]	Fecundity[a]	Population Growth Rate (λ)[b]
Upper	South Carolina	0.90	0.45	0.183	2.288	1.41
	Georgia	0.90	0.45	0.175	1.663	1.27
	Florida	0.90	0.45	0.043	0.531	1.02
Mid	South Carolina	0.70	0.35	0.180	2.250	1.09
	Georgia	0.70	0.35	0.033	0.309	0.75
	Florida	0.70	0.35	0.031	0.383	0.77
Lower	South Carolina	0.62	0.10	0.177	2.213	0.73
	Georgia	0.62	0.10	0.015	0.143	0.62
	Florida	0.62	0.10	0.019	0.235	0.63

[*]Upper, Mid, and Lower Ranges represent the greatest (upper), mean (mid), and lowest (lower) estimates of the metrics used to calculate Fecundity and Population Growth Rate.
[†]Estimates for South Carolina were from the North Inlet-Bulls Island population.
[‡]References: Borowske 2015, pp. 103-104; Roberts et al. 2017, p.122; A. Schwarzer, 2017, pers. comm. and unpublished data.
[§]Period nest survival was calculated using a 24-day nest period (3-day laying period, 11-day incubation period, and 10-day nestling period; Schwarzer & Cox, 2017, p. 9). South Carolina and Georgia rates were calculated using MCestimate (Hunter, et al., 2016b, pp. 137-138; Laskaris, 2016, p. 23) and the Florida rate was calculated using logistic-exposure (A. Schwarzer, 2017, unpublished data).
[a]Fecundity = Period Nest Survival Probability x Mean Number of Nest Attempts Per Breeding Pair x Mean Number of Fledglings per Successful Nest; the Mean Number of Nest Attempts Per Breeding Pair was held constant across Analysis Units and Ranges (mean number of nest attempts = 5) and the values for Mean Number of Fledglings per Successful Nest were the same across Ranges for each Analysis Unit (South Carolina = 2.50, Georgia = 1.90, Florida = 2.47).
[b]Population Growth Rate = Apparent Annual Adult Survival Probability + Apparent Annual First Year Survival Probability x (Fecundity x Proportion of Mean Number of Fledglings per Successful Nest that are Female); the Mean Number of Fledglings per Successful Nest that are Female was the same across Ranges and Analysis Units (0.5).

4.3.2 Current Subspecies Representation

There are currently three genetically distinct representative units of MacGillivray's seaside sparrow across its range: 1) the Headquarters Island population, 2) North Inlet-Bulls Island population, and 3) the Georgia-Florida population. Of the three genetically distinct representative units described above, the Headquarters Island population is the most genetically unique population within the MacGillivray's seaside sparrow range (Woltmann & Hill, 2017, p. 4). The North Inlet-Bulls Island population and the Georgia-Florida population are weakly differentiated from one another (Woltmann & Hill, 2017, p. 1). Preliminary genetic results did not include samples from the ACE Basin population, and therefore, we cannot infer if this population is similar or distinct from the other MacGillivray's seaside sparrow populations.

Additional representative units include three habitat types: two habitat types used by MacGillivray's seaside sparrow in South Carolina and one habitat type used by the subspecies in Georgia-Florida. The Georgia-Florida population breeds in a different marsh habitat type, higher elevation low salt marshes, than the three populations that occur in South Carolina. For the Headquarters Island population in South Carolina, birds occupy only low elevation high marsh habitats. MacGillivray's seaside sparrows in the ACE Basin and North Inlet-Bulls Island populations occur in both low elevation high marsh and managed impoundments. Both natural marsh habitat types experience unrestricted tidal flow, while the managed impoundment representative unit experiences human-controlled hydrologic flows.

The final representative unit for MacGillivray's seaside sparrow is tidal variability, which reflects the subspecies' ability to utilize habitat that experiences a range of tidal fluctuation. Coastal geomorphology along the South Atlantic Bight causes changes in tidal range from lower tidal amplitudes at the edges of the Bight to higher tidal amplitudes at the center of the Bight. This being the case, MacGillivray's seaside sparrows that inhabit natural coastal marshes encounter tides ranging from approximately four to ten feet in elevation, while birds that inhabit managed impoundments do not experience tidal fluctuations. The capacity of MacGillivray's seaside sparrows to adapt to different tidal situations may help birds' improve their nest-site selection behavior when contending with the dueling risks of inundation and predation during the nesting season (Section 2.5).

4.3.3 Current Subspecies Redundancy

MacGillivray's seaside sparrow currently has limited redundancy across its range. The subspecies currently occurs in approximately 73% of its known range in South Carolina, Georgia, and northeast Florida (Figure 4-1, Table 4-1). MacGillivray's seaside sparrow has been extirpated from three and a half counties in northeast Florida: Duval County south of the St. Johns River and St. Johns, Flagler, and Volusia counties. The prevailing hypothesis for this contraction in range is due to the northward expansion of mangroves into previously suitable nesting habitat, although heavy spraying of DDT for mosquito control also may have contributed to the decline (Kale, 1996, pp. 609-610). Based on historical literature, current survey data, and expert judgement, the subspecies is not presumed to be extirpated from any additional locations throughout its range.

There are currently four populations spread throughout the subspecies' range with the exception of the extirpated zone in Florida (Figure 4-1, Table 4-1). The three South Carolina populations are likely stable and have higher resiliency than the Georgia-Florida population. The scattered and patchy breeding distribution of MacGillivray's seaside sparrows in South Carolina has been recognized for nearly a century (Sprunt, 1924, p. 483; Sprunt, 1927, p. 424), and this pattern continues to be reflected in the current distribution found within the state (Hill & Laskaris, 2015, p. 3). The Georgia-Florida population is experiencing a declining trend in resiliency due to low nest survival rates and negative population growth rates as outlined in Section 4.3.1 (and further supported by model simulations in Appendix C). Although there are still four populations spread throughout the subspecies' range, the redundancy is reduced due to all three populations with greater resiliency (the South Carolina populations) being limited to the northernmost third of the currently occupied range, and the downward trend in resiliency for the largest population, Georgia-Florida. In addition, connectivity between the populations is likely low as individuals have high breeding and wintering site fidelity and might only move within a single estuary in their lifetime (Hunter, 2016, p. 163; Shaw, 2012, p. 16-17; A. Given, 2017, unpublished data; W. Post, pers. comm. in Hill & Laskaris, 2015, p. 5).

CHAPTER 5 – FUTURE CONDITIONS AND VIABILITY

We have considered what the MacGillivray's seaside sparrow needs for viability and the current condition of those needs (Chapters 2 and 4), and we reviewed the factors that are driving the current, and future conditions of the subspecies (Chapter 3). We now consider what the subspecies' future condition is likely to be. We apply our future forecasts to the concepts of resiliency, representation, and redundancy to describe the future viability of the MacGillivray's seaside sparrow.

5.1 Introduction

As outlined in Chapter 3 – Factors Influencing Viability, predation and tidal flooding are the primary factors affecting nest success for MacGillivray's seaside sparrow. To predict future resiliency of MacGillivray's seaside sparrow, we used a model to forecast nest success, nest predation, and nest inundation from an observed, historical trend (1975-2015) to the year 2113 for the Georgia analysis unit of the Georgia-Florida population given a range of plausible, future scenarios (Section 5.2). Then, we adapted the model to add increased storm frequency and forecasted nest success when considering the factors of predation, inundation due to sea-level rise, and increased storm frequency (Section 5.2.4). Because of the lack of historical and current population estimates for MacGillivray's seaside sparrow, changes in breeding habitat area were used as a surrogate metric to evaluate the future condition of subspecies. Therefore, we predicted the change in high-quality breeding habitat of MacGillivray's seaside sparrow for the Georgia and South Carolina analysis units from its current condition to years 2025, 2050, 2075, and 2100, and we predicted the future population size of the Georgia analysis unit by projecting changes in the quantity and spatial extent of high quality habitat to years 2025, 2050, 2075, and 2100 (Section 5.3). The models forecasting nest success (demographic factor) and change in high-quality breeding habitat (habitat factor) were developed using similar time frames (current condition to approximately 2100), and therefore, interpretation of model results should consider how these demographic and habitat factors will concurrently influence MacGillivray's seaside sparrow. For example, the forecasting nest success model does not account for how forecasted changes in high-quality breeding habitat may impact future nest success.

Models (Section 5.2 and 5.3.1) were developed as part of a dissertation researching the vulnerability of salt marsh bird populations, including MacGillivray's seaside sparrow, in Georgia to sea-level rise (Hunter, 2016, entire), and these models and scenarios have been published in peer-reviewed journals (Hunter, 2017, entire; Hunter, et al., 2017, entire). Incorporating increased storm frequency into the model for nest success for the Georgia analysis unit (Section 5.2.4), adding a development (SLEUTH) component to the high abundance breeding habitat model for the Georgia analysis unit (Section 5.3.1), and applying the high abundance breeding habitat model to the South Carolina populations (Section 5.3.2) were done specifically for this SSA (E. Hunter, 2017, unpublished data).

5.2 Forecasting Nest Success When Considering the Factors of Predation and Inundation

In the future, sea-level rise will increase the magnitude of regular tidal flooding, and therefore, nesting success will be impacted (Hunter, 2017, p. 459). As further described in Section 2.5, MacGillivray's seaside sparrows face a trade-off in nest site selection based on the concurrent risks of nest predation and nest flooding. Therefore, any future risk to nesting success from inundation due to sea-level rise should also consider predation risks at the same time (Hunter, 2017, p. 459). Climate change will not only affect regular tidal cycles but will affect the frequency and severity of storm events, which may result in further impacts to nesting (Hunter, 2017, pp. 459, 466). MacGillivray's seaside sparrows synchronize their nesting to avoid regular spring tide flooding events. Since storms may occur during any time during a tidal cycle, the unpredictable timing of a storm event can impact MacGillivray's seaside sparrow nesting success (Hunter, 2017, p. 467).

To assess the future viability of MacGillivray's seaside sparrow, we forecasted nest success when considering the factors of predation and inundation due to sea-level rise for the Georgia analysis unit. Also, we adapted the model to forecast nest success when considering increased storm events in addition to the factors of predation and inundation due to sea-level rise. Since sea-level rise (and increased storm frequency) will occur throughout the subspecies' range, and nest-site selection behavior is a behavioral adaption that was observed in a similar habitat type for the Florida analysis unit as in the Georgia analysis unit, the forecast nest success probabilities for the Florida analysis unit are expected to be similar to those for the Georgia analysis unit. Since the modeling efforts (and the three years of data informing them) for the Georgia analysis unit were recently conducted and published just prior to the start of this SSA, and because we only had one year of nest survival data from one South Carolina population, future nest success was not forecast for the South Carolina populations. As stated previously, sea-level rise (and increased storm frequency) is expected in areas where the South Carolina populations occur, but the nest-site selection behaviors observed in the Georgia analysis unit may be different in South Carolina populations which inhabit lower elevation areas of high marsh and impoundments. Therefore, we cannot draw inferences for future nest success of the South Carolina populations based on models run for the Georgia analysis unit.

5.2.1 Model Development

In order to understand the risk to nesting success from predation and inundation due to sea-level rise, an individual-based model of MacGillivray's seaside sparrow nesting behaviors and outcomes was developed (Figure 5-1) (Hunter, 2017, p. 460). A brief description of the model is provided; a full description can be found in Hunter, 2017 (p. 461). Data collected, including estimates of daily nest survival and failure rates, nest height, tidal height, nesting attempts, and breeding season length, from the MacGillivray's seaside sparrow Georgia analysis unit from 2013-2015 were used to calibrate the model (Hunter, 2017, p. 461). The time scale for the model is a breeding season, April 15-July 31, with a daily time step, and nesting female MacGillivray's seaside sparrows were the only model entities used (Hunter, 2017, p. 461). For each model run, 25 nesting females were simulated, and each day, each female's nest was assigned a fate of surviving, being inundated, or being depredated (Hunter, 2017, p. 461). Each female was allowed five nesting attempts, the maximum number of attempts observed in Georgia (Hunter,

2017, p. 463). Parameters (e.g., predation risk) and values (e.g., 0 [low], 1 [high]) developed and used in the individual-based model of MacGillivray's seaside sparrow nesting success are reported in Hunter, 2017 (p. 462). Average daily nest survival, predation, and flooding (inundation) rates and average nest heights across all nests were calculated at the end of each model run (Hunter, 2017, p. 463).
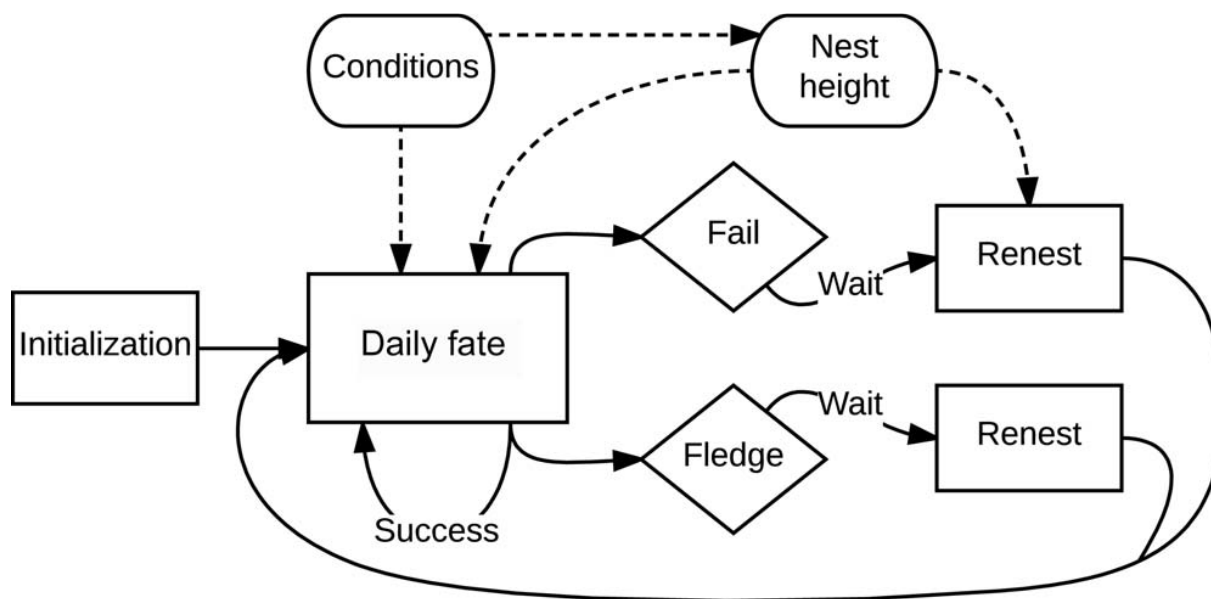
Figure 5-1. Reproduced with permission: "Schematic of an individual-based model of MacGillivray's seaside sparrow nesting success. Boxes are model processes, diamonds are nest fates, and ovals and inline words are parameters or input data. Solid arrows indicate model flow path, and dashed arrows indicate variable effects. The model is initialized at random nest heights for all nesting females. Nests' daily fates are affected by conditions (tidal height and predation risk) and nest height. Each day, nests may fail or succeed, and nests that succeed over the requisite number of days fledge. Females then wait to renest, and previous nest height affects renesting height for females that had nest failure from flooding. Average nest height for a season is also affected by conditions of predation risk." (Hunter, 2017, p. 461).

Two parameters in the model were calibrated to fit empirical data from the MacGillivray's seaside sparrow Georgia analysis unit from 2013-2015. Baseline probabilities of predation and flooding were calibrated to produce realistic outputs (Hunter, 2017, p. 463.). Then, the model was run with calibrated parameter values to ensure daily nest survival rates and nest heights were within 10% of observed values (Hunter, 2017, p. 463). Once calibrated, the model was run to assess two types of scenarios for nest survival rates. The first scenario was historical flood-risks versus higher flood-risk under sea-level rise conditions, and the second scenario was low versus high predation-risk (Hunter, 2017, p. 463).

5.2.2 Scenario Development

Observed, Historical Flood-Risk (1979-2015)
In order to understand the effects of observed, historical flood risk on nest survival, tidal gauge data were downloaded for the breeding seasons 1979-2015, and the number of days with tides over 1.5 m above mean sea level (a tidal amplitude at which most nests are inundated) was calculated to determine each year's flood-risk (Hunter, 2017, p. 463). By regressing flood-risk against years, results indicated that flood-risk has been significantly increasing (P=0.01) over the past 37 years (Hunter, 2017, p. 463), and this increase is likely due to sea-level rise.

Simulated Flood-Risk (2013- 2113) Under Two Sea-Level Rise Scenarios
In order to understand the effects of future flood-risk on nest survival, the observed, historical flood-risk relationship (1979-2015) was extrapolated into the future to create two sea-level rise scenarios of increased flood-risk: 1) flood-risk under moderate sea-level rise and 2) flood-risk under severe sea-level rise (Hunter, 2017, p. 463). For both scenarios, 2013 was used as the baseline year for tidal height. The historical, observed trend of increase in flood-risk was extrapolated linearly for 100 years to develop the flood-risk under the moderate sea-level rise scenario. To develop the flood-risk under the severe sea-level rise scenario, the historical, observed trend of increase was doubled and extrapolated for 100 years. A summary of the two flood-risk scenarios is below; a more-detailed description of the scenarios can be found in Hunter, 2017 (p. 463).

1) Flood-Risk Under Moderate Sea-Level Rise Scenario –
   - Increased all 2013 tidal heights greater than the mean tidal height by 10%.
   - Highest tide was 1.8 m above mean sea level; realistic value of highest tide since this value was observed in the 2012 tidal dataset.
   - Flood risk of 27 days in the breeding season (compared to the historical observed flood risk of 0-12 days).
2) Flood Risk Under Severe Sea-Level Rise Scenario –
   - Increased all 2013 tidal heights greater than mean tidal height by 20%.
   - Highest tide was 1.97 m above mean sea level; value above observed past values.
   - Flood risk of 39 days in the breeding season (compared to the historical observed flood risk of 0-12 days).

Flood-Risk Scenarios
Eleven observed, historical flood-risk scenarios were developed by randomly selecting a year in the 1979-2015 dataset to represent an observed flood-risk value (0-12 days with tides over 1.5 m above mean sea level) (Hunter, 2017, p. 463). With the two simulated flood-risk scenarios (representing sea-level rise), there are a total of 13 flood-risk scenarios.

Adding Predation-Risk to the Flood-Risk Scenarios
In order to understand the effects of predation-risk on nest survival under each flood-risk scenario, two predation-risk scenarios were developed: 1) low predation-risk and 2) high predation-risk (Hunter, 2017, p. 463). A summary of the two predation-risk scenarios is below; a more-detailed description of the scenarios can be found in Hunter, 2017 (p. 463).

1) Low Predation-Risk Scenario –
   - Used the lowest average daily predation rate (0.05) from empirical data collected during 2013-2015. Data collected during 2014 had the lowest average rate.
2) High Predation-Risk Scenario –
   - Used the average values from two years of high average daily predation rates (0.09 and 0.15) from empirical data collected during 2013-2015. Data collected from 2013 and 2015 had the highest average rate.

Nest-Height Selection Behavior
MacGillivray's seaside sparrows have adapted behaviorally to balance the trade-off in risk from predation and flooding and therefore, will shift nest-site placement along a nest height gradient (Hunter, 2017, p. 460). In order to understand the effects these behaviors have on outcomes of the scenarios, all scenarios were run with nest-height selection behaviors and without nest-height selection behaviors in response to flooding and predation events (Hunter, 2017, p. 464).

5.2.3 Model Run and Statistical Analysis

Each Flood- and Predation-Risk scenario combination was run 100 times, and outputs of daily nest survival, flooding, and predation rates were calculated (Hunter, 2017, p. 463). A brief description of the statistical analysis is below; a more-detailed description can be found in Hunter, 2017 (p. 464). Random forests were used to analyze the contribution of predation-risk and flood-risk to nest survival (Hunter, 2017, p. 464). Change in nest survival, flooding, and predation probabilities as functions of predation- and flood-risk also were measured, and these probabilities with nest-height selection behavior were compared to the models without nest-height selection behaviors (Hunter, 2017, p. 464).

5.2.4 Individual-Based Model Adaptation

The individual-based model of MacGillivray's seaside sparrow nest survival (summarized above; Hunter, 2017, pp. 460-464) was adapted for this SSA to include increased storm events. In order to understand the effects of future flood-risk on nest survival, flood-risk from sea-level rise was simulated as an increase in the magnitude of "regular" flooding – that is, flooding during spring tide events that occur approximately every 28 days. Two years of tidal records that had only "regular" tides (no high tides between spring tide events that would likely be storm-caused) were used, and therefore, two flood-risk scenarios were developed based on the number of flood-risk days (i.e., days with tides >1.5 m above mean sea level which typically cause nest failure; Hunter, et al., 2016b, p. 138; Hunter, 2017, p. 463):

1) Low "Regular" Flood-Risk Scenario –
   - Used tidal records from 2010 where there were 0 flood-risk days.
2) High "Regular" Flood-Risk Scenario –
   - Used tidal records from 2013 where there were 12 flood-risk days.

In order to understand the effects of future storm events on nest survival, a "storm" (a single day with a 2 m tide) was inserted in between two spring tides in each flood-risk scenario. This storm could occur either early in the breeding season (mid-May) or late in the breeding season (early

July). Also, two predation-risk scenarios, low- and high predation-risk, were added to each flood-risk scenario (Hunter, 2017, p. 463). All models included nest-height selection behaviors. Nesting survival rates from each model output were compared among the different scenarios.

5.2.5 Results and Discussion

The following is a summary of the individual-based model results; a full description can be found in Hunter, 2017 (pp. 464-465). When running the model with randomly-selected years (1979-2015) from the historical, observed flood-risk scenarios, results indicate that predation-risk had a greater influence (17 times greater) than flood-risk on daily nest survival rates (Hunter, 2017, p. 465). With the exception of the flood-risk under the severe sea-level rise scenario, flood-risk had limited to no effect on predation probability, while predation-risk had a similar effect on flooding probability compared to flood-risk (Figure 5-2 A; Hunter, 2017, pp. 465-466).

Nest-height selection behaviors from the model had small positive effects on daily nest survival rates across all historical, observed flood-risk scenarios, except for the 11-day flood-risk scenario under high predation-risk (Figure 5-2 C) (Hunter, 2017, pp. 465-466). This flood-risk scenario also had higher flooding probabilities than expected based on the trend of other flood-risk scenarios (Figure 5-2 A) (Hunter, 2017, p. 465). Removing nest-height selection behaviors had a greater overall effect on flooding probability than on predation probability (Figures 5-2 A, 5-2 B) (Hunter, 2017, p. 464). Flooding probability under low predation-risk scenarios was the most affected, with a 26.1% increase in flooding probability on average when birds did not alter their nest-height selection behavior following nest failure (Hunter, 2017, pp. 464-465). Under historical, observed flood-risk scenarios, flooding probability increased by 23.5% in scenarios with greater flood-risk days and by 21.4% in scenarios with fewer flood-risk days when selection behaviors were removed from the model (Figure 5-2 A). (Hunter, 2017, pp. 464-465).

For the two simulated flood-risk under sea-level rise scenarios, predation-risk was 6.8 times more important than flood-risk in determining daily nest survival (Hunter, 2017, p. 464). As expected, daily flooding probability increased from the historical, observed flood-risk years to the moderate sea-level rise scenario and was highest under the severe sea-level rise scenario (Figure 5-2 A), which caused declines in daily nest survival probability (Figure 5-2 C) (Hunter, 2017, pp. 464-465). The effects of nest-height selection behaviors from the model resulted in a lower flooding probability for low predation-risk as compared to low predation-risk without selection behaviors and high predation risk with or without behaviors modeled (Figure 5-2 A). Since suitable nesting habitat will likely be lost prior to such high levels of flooding occurring, the simulated flood-risk under the severe sea-level rise scenario is highly unlikely to occur (Hunter, 2017, p. 465). The simulated sea-level rise scenarios were modeled as an increase in the number of days with tides over 1.5 m above mean sea level, but the effect of sea-level rise on high quality breeding habitat was not modeled here (see Section 5.3 for this modeling).

With both flood- and predation-risk very low, the baseline daily nest survival rate was 0.93 ± 0.01 (mean ± SD; Hunter, 2017, p. 464). Daily nest survival was reduced from the baseline to 0.83 ± 0.03 with increasing (high) predation-risk under the historical, observed 0-day flood-risk scenario (Figure 5-2 C; Hunter, 2017, p. 464). When compared to changes in nest survival under the simulated flood-risk under severe sea-level rise scenario, daily nest survival was reduced to

0.84 ± 0.02 from baseline, almost equal to the change caused by increasing predation-risk (Figure 5-2 C; Hunter 2017, p. 464). It is important to note that the difference in nest survival probability ($\delta = 0.10$) between high and low predation-risks under historical, observed flood-risk is the same as the difference in nest survival probability ($\delta = 0.09$) that would occur between a low "regular" flood-risk year and the flood-risk under severe sea-level rise (Figure 5-2 C; Hunter 2017, p. 465). This high level of predation-risk is currently being observed in the Georgia-Florida population (Table 4-3). In addition, the severe sea-level rise scenario is unlikely to occur since suitable nesting habitat will already be lost before this level of extreme flooding can impact nests (Hunter, 2017, p. 465). Therefore, current observed predation levels pose a greater direct risk to nesting success than future sea-level rise is likely to cause (Hunter 2017, p. 465).
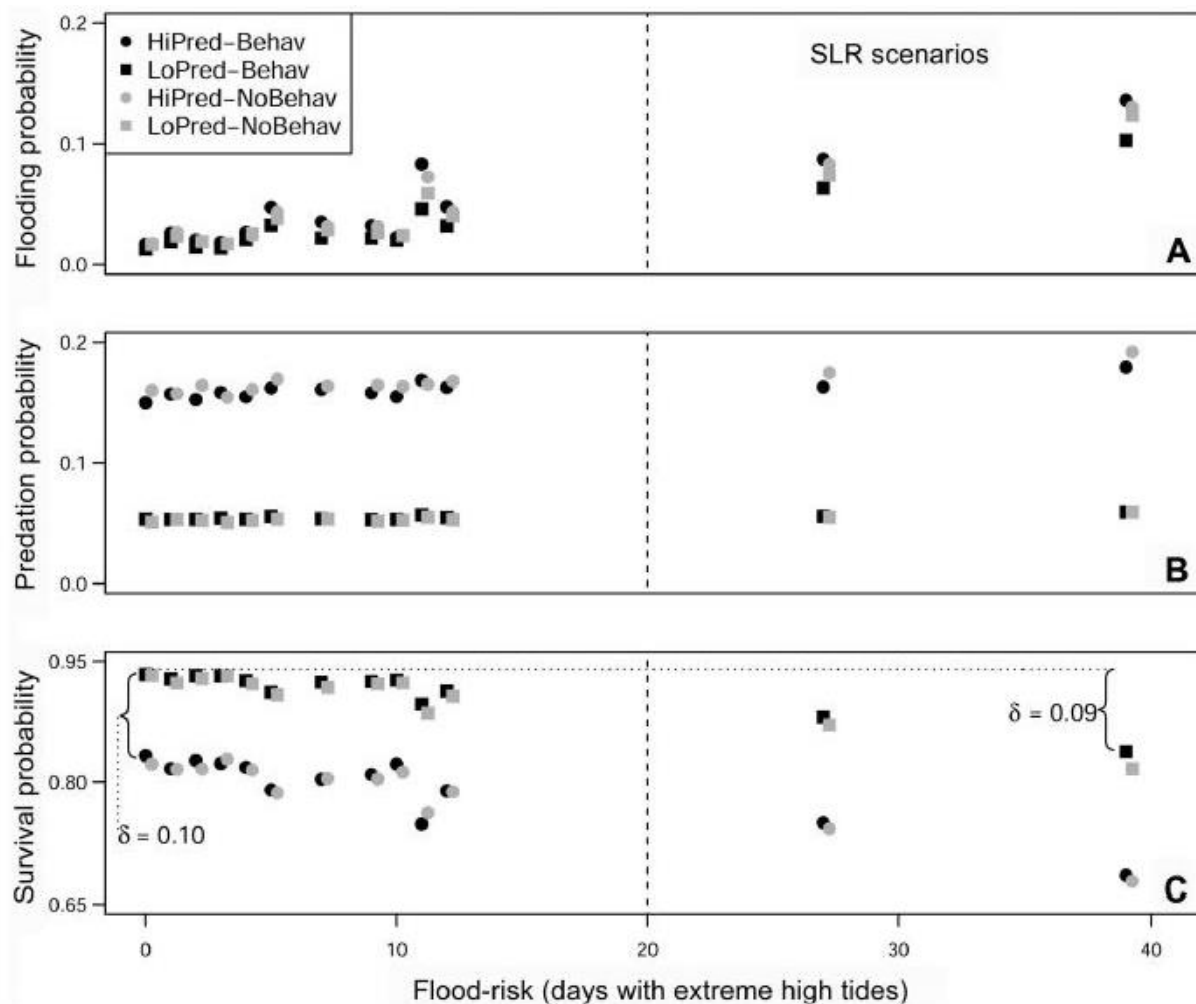
Figure 5-2. Flooding (A), predation (B), and nest survival (C) probabilities (averaged across 100 simulations for each point) for MacGillivray's seaside sparrow (*Ammodramus maritimus macgillivraii*) nests under eleven observed, historical flood-risk scenarios (left of dotted line) and two simulated flood-risk under sea-level rise scenarios (right of dotted line), with high (circles) and low (squares) predation-risk. "Black points are mean probabilities from models including nest-height selection behaviors, and gray points are mean probabilities from models without those behaviors. Brackets in C indicate the difference in nest survival rate between scenarios of lowest and highest predation (left bracket) and flooding (right bracket) risks at the lowest risk for the other threat." Reproduced with permission (Hunter, 2017, p. 465).

When the individual-based model was adapted to include a storm event at different times during the breeding season, results suggested that late-season storms had little to no effect on nest survival rates, but early-season storms had a substantial negative effect on nest survival, for both low and high "regular" flood-risk years (Figure 5-3). This difference between early- and late-season is likely due to a difference in probability of nesting – all females nest in mid-May, but by early July, a substantial proportion of females stop making nesting attempts. If all females lose their nests to a storm-based flood in mid-May, most will likely lose their subsequent nests to either predation (due to changes in nest height) or to the next spring tide; whereas, a late-July storm-based flood is not likely to result in the same ecological cascade of events that are predicted to result in future nest loss (because peak nesting ends in mid- to late-July). However, the occurrence of flood-causing storms is much more likely later in the breeding season when the tropical storm and hurricane season begins (technically, June 1, but storms become more likely later in the summer). A storm early in the breeding season that can cause devastating flooding without the aid of a spring tide, is unlikely.
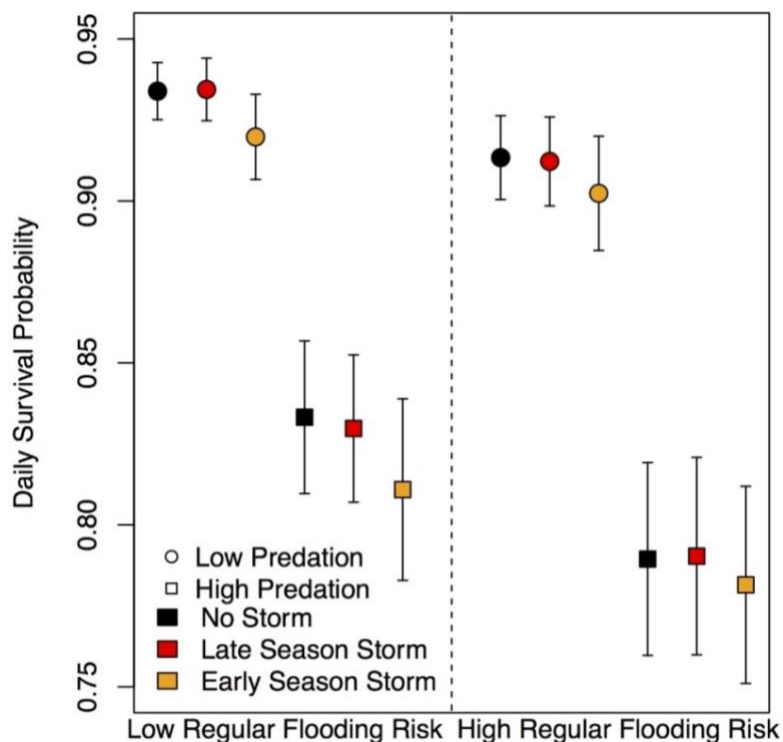
Figure 5-3. Daily MacGillivray's seaside sparrow (*Ammodramus maritimus macgillivraii*) nest survival rates as affected by storm-flooding and predation. Estimates are means and standard deviations across 100 simulations. The left panel shows estimates when there is a low "regular" flood-risk (0 days) during spring tides; the right panel shows estimates for high "regular" flood-risk (12 days). Circles are low predation-risk simulations and squares are high predation-risk simulations. Colors represent timing of storm events: no storm (black), late-season storm (red), and early-season storm (orange). Compare to Figure 5-2.

5.3 Forecasting Change in Breeding Habitat from Inundation Due to Sea-Level Rise

As summarized in Section 5.2, the effect of sea-level rise on MacGillivray's seaside sparrow nest success was forecasted, and current observed predation levels pose as great a risk to nesting success as future sea-level rise is likely to cause (Hunter, 2017, pp. 465-466). In addition, suitable nesting habitat will likely be lost to inundation due to sea-level rise before the effects of severe sea-level rise (high levels of flooding) on nest success are realized (Section 5.2, Figure 5-2). Therefore, it is important to understand how breeding (nesting) habitat will change from inundation due to sea-level rise in the future.

To assess the future viability of MacGillivray's seaside sparrow, we projected the change in high abundance breeding habitat from sea-level rise for the Georgia-Florida population and South Carolina populations.

5.3.1 Georgia-Florida population

5.3.1.1 Model Development

A brief description of the models developed is provided; a full description of the Bayesian and sea-level rise models developed for MacGillivray's seaside sparrow can be found in Hunter, et al., 2017 (pp. 22-24). First, in order to estimate the MacGillivray's seaside sparrows' response to environmental factors, current landscape gradients were generated using raster map outputs from the Sea Level Affecting Marshes Model (SLAMM) and measured within the 200-m radius circle around each sampling point (Hunter, et al., 2017, p. 22). In SLAMM, the 2007 National Wetland Inventory (NWI) wetland cover and relative elevation raster outputs from 2007 were used for the current time period (Hunter, et al., 2017, p. 22). Landscape gradients of distance to developed and forested areas, edge density, elevation, patch area, the proportion of brackish marsh (an indicator of salinity), and the proportion of the landscape that is marsh (an indicator of channel size) were used. A Bayesian statistical model was developed to identify the current landscape gradients that influence occupancy and abundance of MacGillivray's seaside sparrows in the Georgia analysis unit (Hunter, et al., 2017, p. 22). This model also was used to estimate the current MacGillivray's seaside sparrow population size in the Georgia analysis unit.

In order to estimate loss of high abundance breeding habitat from inundation due to sea-level rise for the Georgia analysis unit, projections of landscape gradients from SLAMM were used to estimate the potential effects of sea-level rise on habitat availability and MacGillivray's seaside sparrow population size in Georgia (Hunter, et al., 2017, entire). SLAMM projections for four time periods (2025, 2050, 2075, and 2100) were used (Hunter, et al., 2017, p. 22). This modeling was used to examine the change in predicted population size and change in high abundance breeding habitat (defined as the top 25% of the range of predicted bird abundance values) in response to a sea-level rise scenario of a 1 m rise from 2007 to 2100 (Hunter, et al., 2017, pp. 22, 24).

In addition, increased development near salt marshes reduces habitat availability, likely because MacGillivray's seaside sparrows avoid marshes close to developed areas (Hunter, et al., 2016a, p. 3; Hunter, et al., 2017, p. 24). Moreover, sea-level rise is expected to push available habitat

further inland (closer to developed areas), then high abundance breeding habitat for MacGillivray's seaside sparrows could be "squeezed" between rising sea levels and hard physical barriers (many of which are from human infrastructure). To represent the increasing risk that urbanization poses to MacGillivray's seaside sparrow habitat, outputs from the SLEUTH urban growth model (Terando, et al., 2014, entire) were used in addition to the SLAMM outputs to project changes to high abundance breeding habitat as outlined above.

This modeling was performed for the Georgia analysis unit of the Georgia-Florida population, and not for the Florida portion of the population. The Florida analysis unit was not part of the original study, nor was MacGillivray's seaside sparrow data from Florida available at the time of modelling. Since the Georgia-Florida population is considered one contiguous population of MacGillivray's seaside sparrow and individuals use the same habitat type, the results for the Georgia analysis unit are expected to be similar for the Florida analysis unit.

5.3.1.2 Results and Discussion

Distance to development, distance to forested areas, and proportion brackish significantly influenced the current occupancy of MacGillivray's seaside sparrows in the Georgia analysis unit (Hunter, et al., 2016a, p. 4; Hunter, et al., 2017, pp. 22-24). Distance to development and forested areas both had positive effects, meaning marshes further away from development and forests had higher occupancy, and proportion brackish had a negative effect (sites with less brackish marsh had higher occupancy; Hunter, et al., 2016a, p. 4; Hunter, et al., 2017, pp. 22-24). Elevation and proportion brackish significantly influenced MacGillivray's seaside sparrow current abundance (Hunter, et al., 2016a, p. 5; Hunter, et al., 2017, pp. 22-24). Elevation had a positive effect, sites with higher elevations had higher densities, and proportion brackish again had a negative effect (Hunter, et al., 2016a, p. 5; Hunter, et al., 2017, pp. 22-24). Therefore, MacGillivray's seaside sparrows in the Georgia analysis unit were more likely to be found in higher elevation areas of low salt marsh far from forests and development (Hunter, et al., 2016a, p. 5). At the time of the study, MacGillivray's seaside sparrow population size in Georgia was estimated to be 32,800 birds (95% CI: 25,200-40,400; Hunter, et al., 2016a, p. 5).

Using the estimated effects of the landscape gradients and SLAMM projections, the area of high abundance breeding habitat and population size showed overall declines from current to the year 2100 (Table 5-1, Figures 5-4 and 5-5; Hunter, et al., 2016a, p. 6; Hunter, et al., 2017, p. 26). High abundance breeding habitat is predicted to increase by 7% by the year 2025, and then by 2050, 27% of the current area is predicted to be lost (Table 5-1). MacGillivray's seaside sparrow population size was forecasted to decline by 54% while the current area of high abundance breeding habitat is predicted to decline by over 80% by the year 2100 (Hunter, et al., 2016a, p. 6; Hunter, et al., 2017, p. 26). The loss of over half of the MacGillivray's seaside sparrow population in Georgia by 2100 can be attributed to loss of high abundance breeding habitat as opposed to across the board losses of salt marshes (Hunter, et al., 2016a, p. 6); in fact, SLAMM predicts only a 6% decline in overall salt marsh area in Georgia by 2100 (Hunter, et al., 2015, p. 1534). The rapid rates of decline are predicted to occur between the years 2025 and 2050 when sea-level rise rates start to accelerate (Hunter, et al., 2016a, p. 6).

Table 5-1. Percent change in high abundance breeding habitat for MacGillivray's seaside sparrow (*Ammodramus maritimus macgillivraii*) in the Georgia analysis unit. Change is estimated from a baseline high abundance breeding habitat designated using the top 25% of predicted abundances for the site. Change is estimated using projections from the Sea Level Affecting Marshes Model (SLAMM) version 6.0 and state-specific models of seaside sparrow habitat use.

| Population | Year | | | |
Analysis Unit	2025	2050	2075	2100
Georgia-Florida				
Georgia	+7	-27	-41	-81
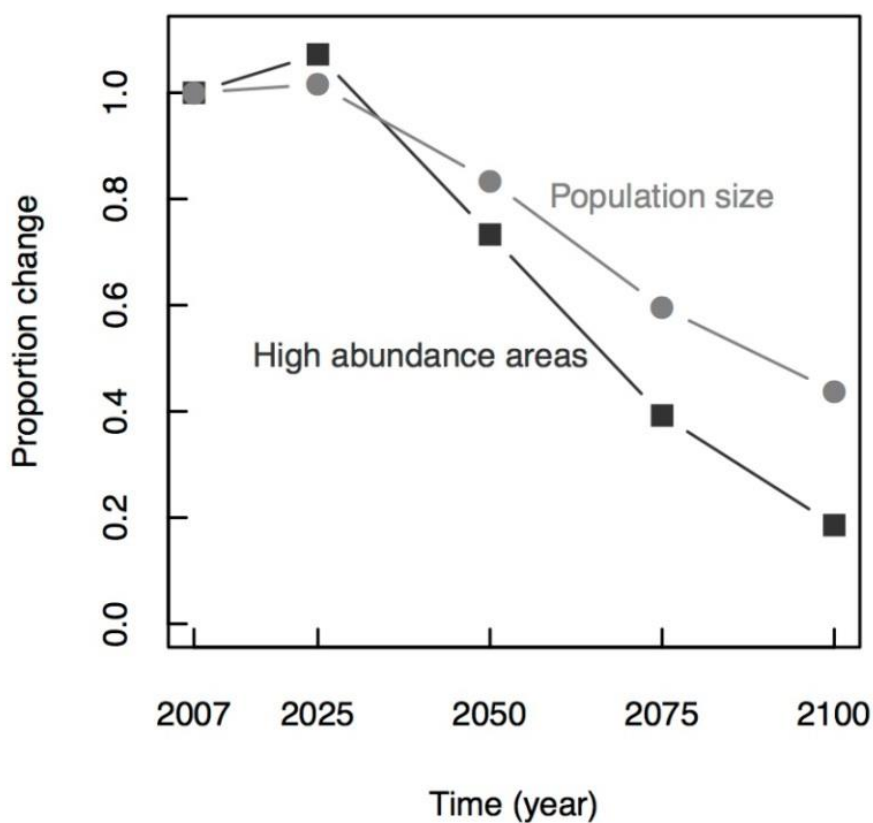
Figure 5-4. Predicted changes in population size and the amount of high abundance breeding habitat for MacGillivray's seaside sparrow (*Ammodramus maritimus macgillivraii*) in coastal Georgia marshes due to sea-level rise. Reproduced with permission (Hunter, et al., 2016a, p. 6).
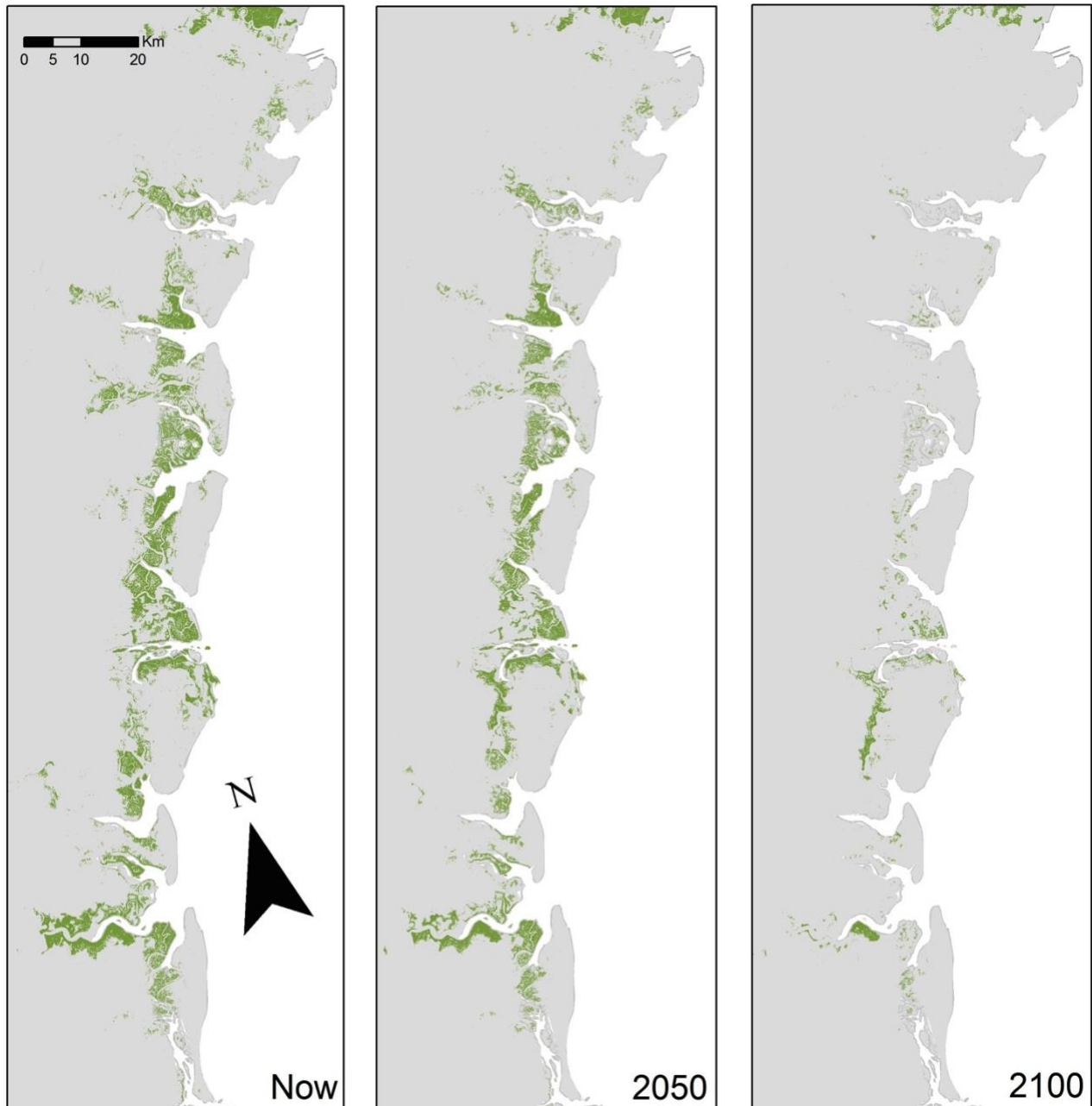
Figure 5-5. Predicted change in the spatial extent of high abundance breeding habitat for MacGillivray's seaside sparrow (*Ammodramus maritimus macgillivraii*) in coastal Georgia marshes due to sea-level rise. Reproduced with permission (Hunter, et al., 2016a, p. 7).

When urban growth projections were included in the model, there was no difference in the estimated loss of high abundance breeding habitat compared to the results of the SLAMM-only model. This is likely because new development is expected to occur at the margins of existing development, and MacGillivray's seaside sparrows are already occupying areas further away from existing development (Hunter, et al., 2017, pp. 24-25). Therefore, increasing development does not have as great an impact to MacGillivray's seaside sparrow habitat as sea-level rise. However, if increasing development boosts predator populations that then make their way into

the subspecies' habitat, urbanization may become one of the primary factors influencing the viability to populations.

The predicted changes in population size and high abundance breeding habitat were performed on the Georgia analysis unit and not the Florida analysis unit. Since the Georgia-Florida population is considered one contiguous population of MacGillivray's seaside sparrow and have similar habitat relationships, the high abundance breeding habitat loss estimates for the Florida analysis unit are expected to be similar to those for the Georgia analysis unit.

5.3.2 South Carolina Populations

5.3.2.1 Model Development

In order to estimate habitat loss from inundation due to sea-level rise for the South Carolina populations, projections from SLAMM were used to estimate the potential effects of sea-level rise on habitat availability in South Carolina. SLAMM projections were available for five NWRs in South Carolina (north to south): Waccamaw NWR, Cape Romain NWR, ACE Basin NWR, Pinckney Island NWR, and Savannah NWR. Estimates of the effects of environmental features (landscape gradients) on MacGillivray's seaside sparrow abundance were used to delineate present-day high abundance breeding habitat (outlined above in Scenario 5.4.1 and following methods in Hunter, et al., 2016a), then the SLAMM projection for the year 2100 was used to determine how much habitat would be lost (or gained) by that year. Predicted change in high abundance breeding habitat from its current condition to years 2025, 2050, 2075, and 2100 was also calculated. SLAMM projections of elevation change were not available, so abundance relationships to elevation were not used in these calculations.

5.3.2.2 Results and Discussion

Higher MacGillivray's seaside sparrow abundance was associated with greater distances from developed areas, lower elevations, and lower salinity marshes. The relationship between distance from developed areas (positive effect meaning marshes further away from developed areas are more suitable habitat) is the same as the relationship found in Georgia, but the effects of elevation and the proportion of brackish marsh (positive effect meaning sites with more brackish marsh are more suitable) are the opposite from those in Georgia. These results indicate that MacGillivray's seaside sparrows in South Carolina are likely using different habitats than MacGillivray's seaside sparrows in Georgia. Therefore, MacGillivray's seaside sparrows in South Carolina are more likely to be found in lower elevation brackish marshes (areas of high marsh) far from development. There was too much uncertainty in the state-wide population size estimate for MacGillivray's seaside sparrow in South Carolina as to be informative (see Appendix B for further information).

Using the estimated effects of environmental features (landscape gradients) and SLAMM projections to estimate change in high abundance breeding habitat, results showed large losses of habitat by the year 2100 (Figure 5-6). Two of the three NWRs that contain surveyed populations (Waccamaw NWR and Cape Romain NWR = North Inlet-Bulls Island population) had the highest estimated losses at 88% and 77%, respectively, by 2100 (Figures 5-6 and 5-7; Table 5-2);

by the year 2050, a loss of 44% and 30% in high abundance breeding habitat was predicted. Whereas, the ACE Basin NWR which contains the ACE Basin population had a low loss rate of 11% by 2100 (and actually increases in habitat area in the time periods before 2100; Table 5-2, Figure 5-6). For Pinckney Island and Savannah NWRs, which do not support defined breeding populations but may support individual sparrows at low densities, estimated habitat loss rates were 15% and 4% by the year 2050, respectively, and 50% by the year 2100 (Table 5-2). The SLAMM projections did not cover the Headquarters Island population; however, this population occupies tidally-influenced high marsh and is equidistant from Cape Romain NWR and ACE Basin NWR, which both show projections of high abundance breeding habitat loss by 2100. Therefore, it is reasonable to infer that the Headquarters Island population also will experience a loss in high abundance breeding habitat due to sea-level rise.

Given the close proximity of Savannah and Pinckney Island NWRs to Georgia, the landscape gradient features of any MacGillivray's seaside sparrow individuals in these areas may be more similar to the Georgia-Florida population. If that were the case, high abundance breeding habitat loss rates at these refuges could be as high as 80% by the year 2100 (see Section 5.3.1.2 for Georgia-Florida habitat loss estimates). Despite substantial uncertainty in the habitat relationships for South Carolina, and differences in occupancy and abundance estimates between South Carolina and Georgia, the loss estimates for high abundance breeding habitat are similar for the South Carolina populations to those for the Georgia-Florida population.

Table 5-2. Percent change in high abundance breeding habitat for MacGillivray's seaside sparrow (*Ammodramus maritimus macgillivraii*) in five National Wildlife Refuges in South Carolina. Change is estimated from a baseline high abundance habitat area designated using the top 25% of predicted abundances for the site in the early 2000s (baseline year varies among the sites). Change is estimated using projections from the Sea Level Affecting Marshes Model (SLAMM) version 6.0 and state-specific models of MacGillivray's seaside sparrow habitat use.

National Wildlife Refuge	Year			
	2025	2050	2075	2100
Waccamaw	- 11	- 44	- 68	- 88
Cape Romain	- 2	- 30	- 39	- 77
ACE Basin	+ 11	+ 7	+ 12	- 11
Pinckney Island	- 8	- 15	- 49	- 50
Savannah	- 1	- 4	- 20	- 52
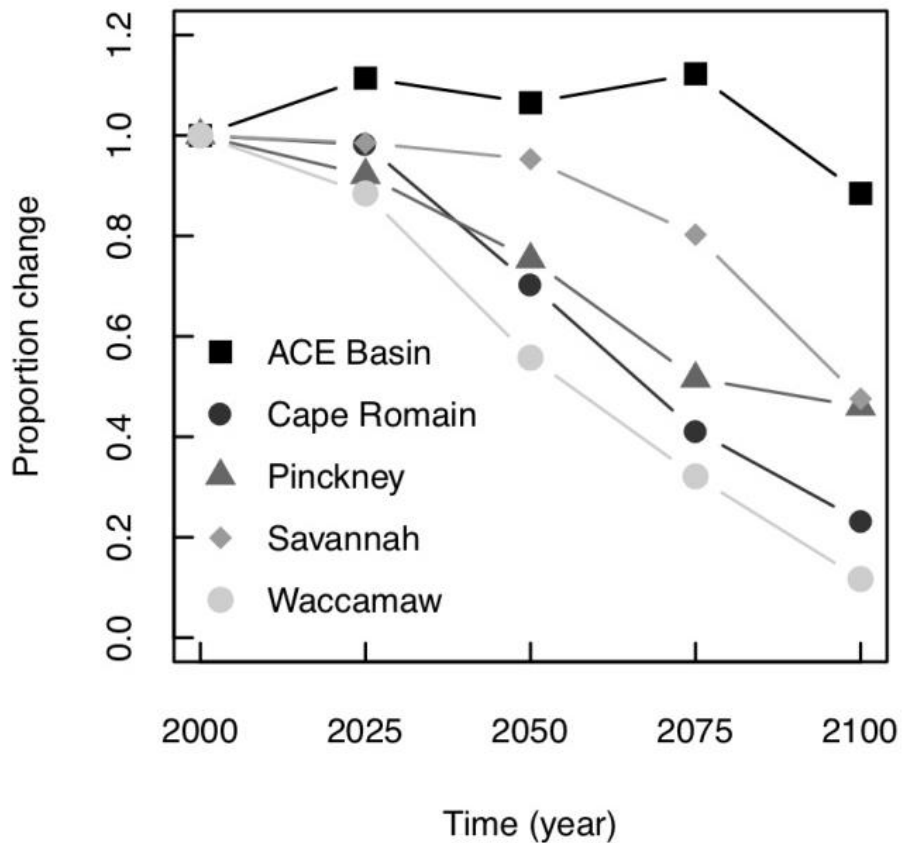
Figure 5-6. Predicted changes in the amount of high abundance breeding habitat for MacGillivray's seaside sparrow (*Ammodramus maritimus macgillivraii*) in five National Wildlife Refuges in South Carolina (north to south): Waccamaw, Cape Romain, ACE Basin, Pinckney Island, and Savannah. Note: Refuges had different "present day" starting points in the analysis, but the year 2000 was used here for consistency.
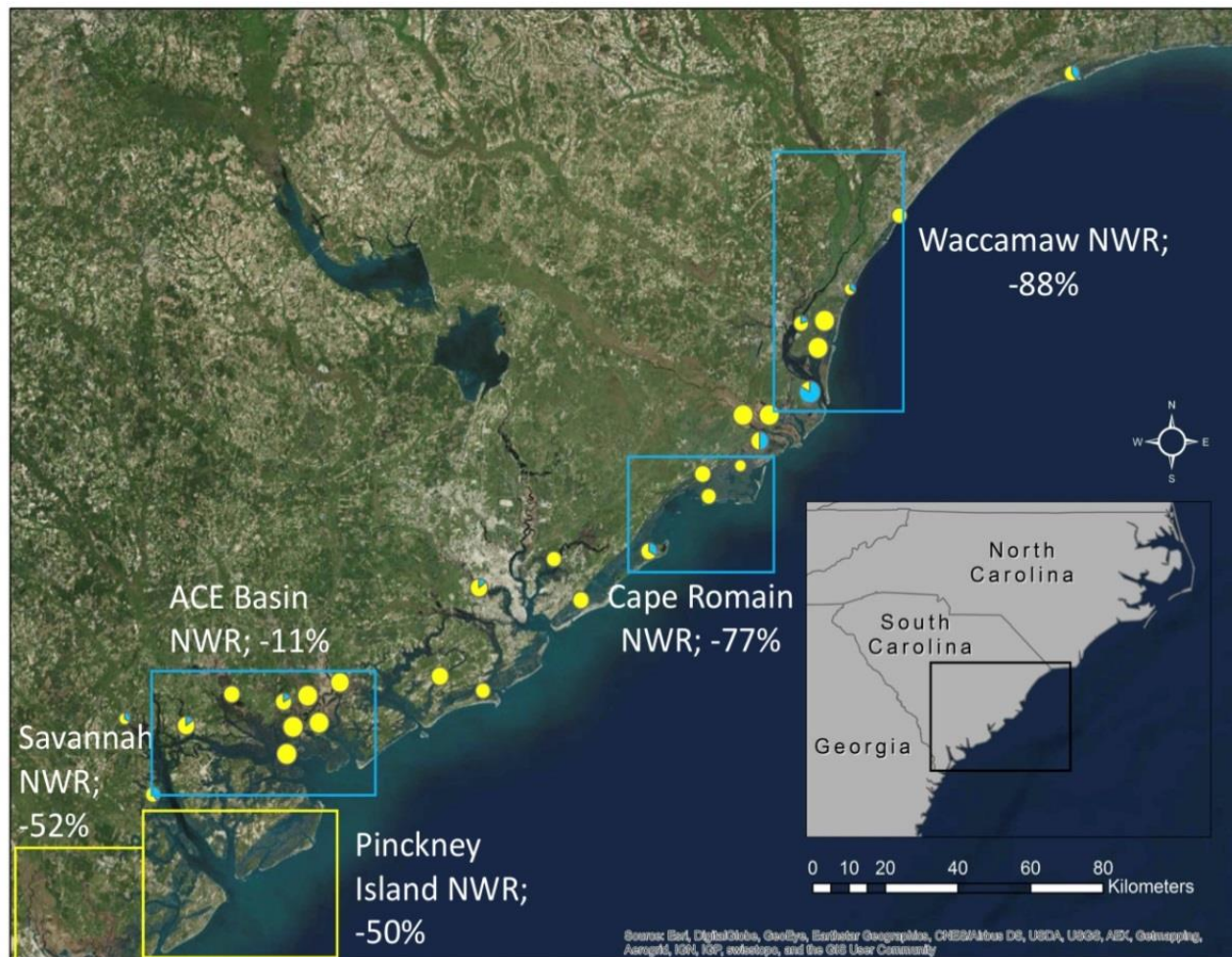
Figure 5-7. The change in MacGillivray's seaside sparrow (*Ammodramus maritimus macgillivraii*) habitat area based on estimated habitat relationships and Sea Level Affecting Marshes Model (SLAMM)-projected marsh change. Yellow circles indicate the number of sites surveyed and blue shading indicates the proportion of surveyed sites occupied during breeding season surveys for the South Carolina status assessment (Hill & Laskaris, 2015, p. 17). Boxes indicate the extent of the SLAMM study area for each National Wildlife Refuge (NWR): blue boxes contain two total MacGillivray's seaside sparrow breeding populations (Cape Romain and Waccamaw NWRs contain portions of the North Inlet-Bulls Island population, and ACE Basin NWR contains the ACE Basin population) and yellow boxes do not contain any defined breeding populations. Percentages indicate the estimated percent change in high abundance breeding habitat from present day to 2100.

5.4 Summary of Future Conditions and Viability based on Resiliency, Representation, and Redundancy

The models forecasting changes in nest success (demographic factor) and high-quality breeding habitat (habitat factor) were developed using similar time frames, present day to approximately 2100. The forecasting nest success model does not account for how forecasted changes in high-quality breeding habitat may impact future nest success. However, these demographic and habitat factors are likely occurring concurrently, and therefore, the interpretation of model results should be considered collectively.

5.4.1 Future Resiliency

Current daily nest survival for the Georgia analysis unit is 86.7% with the Florida analysis unit being 86.5% (Table 4-3), and in turn, most of the current population growth rates estimated indicate that the Georgia-Florida population is declining (i.e., population growth rate < 1.0) (Table 4-4, Appendix C). The individual-based model of MacGillivray's seaside sparrow estimated a baseline daily nest survival rate of 0.93 under low predation-risk and historical, observed 0-day flood-risk scenario, which is higher than the currently observed daily nest survival rate (Figure 5-2 C) (Hunter, 2017, p. 464). Under the same flood-risk scenario but with high predation-risk, the daily nest survival rate is reduced to 0.83, which is closer to the current observed daily nest survival rate. This high level of predation-risk is currently being observed in the Georgia-Florida population. As stated previously, the model results found that predation-risk had a greater influence on daily nest survival rate than flood-risk, even when flood-risk was simulated to increase with sea-level rise (Hunter, 2017, p. 465). In the future scenarios of sea-level rise, daily nest survival was reduced to less than 80% under high predation-risk (Figure 5-2 C). MacGillivray's seaside sparrow nest-height selection behaviors had an overall positive effect on nest survival, and therefore, in terms of nest success, these behaviors will likely contribute at some level to the Georgia-Florida population's resiliency to predation and sea-level rise (Hunter, 2017, p. 466). Little is known about the variability in predation-risk between breeding seasons, and how changes in the frequency of high and low predation-risk impact the capacity of nest-height selection behaviors to adapt and continue contributing to MacGillivray's seaside sparrow resiliency (Hunter, 2017, p. 466). Results from the modelling that incorporated increased storm frequency suggest that storms early in the breeding season had a substantial negative effect on nest survival under both flood-risk scenarios. However, results further indicated that impacts to nesting success from increased storm frequency were far outweighed by the impacts from present-day levels of predation (Hunter, 2017, p. 465).

As mentioned in Section 4.3.1, low nest survival is a leading driver of population declines (Donovan & Thompson III, 2001, entire), and under the current condition, we determined that low nest survival is a major factor in the downward trend for MacGillivray's seaside sparrow in the Georgia-Florida population. Therefore, in the future, continued reductions in daily nest survival will lead to lower period nest survival resulting in even larger negative population growth rates (and accompanying declines in resiliency) for the Georgia-Florida population of MacGillivray's seaside sparrow. When considering the effect of sea-level rise on predator communities, even less is known about how predator populations will respond to changes in tidal and habitat conditions. However, it is unlikely that MacGillivray's seaside sparrow overall predation-risk will decrease as a result of sea-level rise, because many nest predators (e.g., raccoons, grackles) are food generalists that use aquatic, terrestrial, and urban (human) habitats to supplement their diet (Hunter, 2017, p. 466), so predator abundance is not expected to decline. In addition, sea-level rise may lead to a greater concentration of predators in MacGillivray's seaside sparrow breeding areas during the highest high tides.

MacGillivray's seaside sparrows are projected to lose the majority of their high abundance breeding habitat across the subspecies range by 2100, from inundation due to sea-level rise. Although SLAMM projections to 2100 include substantial uncertainty and imprecisely modeled geologic processes (Chu-Agor, et al., 2011, entire; Clough, et al., 2010, entire), it is clear that

some large portion of MacGillivray's seaside sparrow habitat will be lost simply due to the subspecies' relationship with relative elevation (which is directly related to sea level). Studies of population viability in other seaside sparrows have shown that substantially less habitat loss from sea-level rise than what is predicted for MacGillivray's seaside sparrow could still result in an increased population extinction probability by more than 50% from present day (Shriver & Gibbs, 2004, entire).

The large projected declines in habitat availability will reduce the ability of three MacGillivray's seaside sparrow populations to withstand stochastic events, thereby reducing resiliency. In South Carolina, the North Inlet-Bulls Island population is projected to lose 30% to 44% of its high abundance breeding habitat by 2050, and 77% to 88% by 2100, while the ACE Basin population is projected to gain 7% by 2050, but lose 11% by 2100. Based on the location of the Headquarters Island population (approximately 8 miles from the Atlantic Ocean and located equidistant to Cape Romain NWR and ACE Basin NWR) and the tidally-influenced natural marsh that the subspecies uses there, we infer that the Headquarters Island population also will experience a loss in high abundance breeding habitat from sea-level rise. The fourth population, Georgia-Florida, may lose on the order of 80% of its high abundance habitat by 2100 (27% by 2050). In addition, the MacGillivray's seaside sparrow's current population size for the Georgia analysis unit was forecasted to decline by 54% by the year 2100, from 32,800 birds to 15,088 birds, which was attributed to the loss of high abundance breeding habitat (Hunter, et al., 2016a, p. 6). However, as mentioned above, this habitat projection modeling does not take into account the effects of sea-level rise on nest success, which will likely contribute to greater proportional declines in population size at a faster rate.

Estimated future reductions in nest survival, compounded with future projections of habitat loss, will exacerbate the current declining trend in resiliency for the Georgia-Florida population. For South Carolina populations analyzed, the future projections of high abundance breeding habitat loss will likely result in population changes from stable to declining population growth, thereby reducing their future resiliency.

5.4.2 Future Representation

In the current condition, there are three genetically distinct representative units of MacGillivray's seaside sparrow identified across its range. With the loss of high abundance breeding habitat across the range of the subspecies, these genetically distinct units will likely remain into the future but with reduced resiliency for all three representative units (Headquarters Island population, North Inlet-Bulls Island population, and Georgia-Florida population). The South Carolina populations remain isolated with limited to no connectivity as already represented by their current distribution across the state. The Georgia-Florida population may become isolated over time with habitat patches becoming increasingly fragmented under sea-level rise and habitat quality diminishing from increasing isolation.

MacGillivray's seaside sparrows in the Georgia-Florida population use higher elevation low salt marsh habitats far from development and forests, while the South Carolina populations are more likely to be found in lower elevation high salt marshes or impoundments also far from development. Despite this habitat variability, there are projected declines of high abundance

breeding habitat throughout the subspecies range for all three habitat variability representative units. Under moderate and severe sea-level rise scenarios, impoundments may provide remaining habitat for MacGillivray's seaside sparrow; however, impoundments are costly to maintain, especially when impacted by large storm events, and their present management does not specifically include MacGillivray's seaside sparrow. Therefore, the reliability of managed impoundments to support MacGillivray's seaside sparrow in the future is uncertain.

Tidal regimes and increased storm frequency and severity are projected to change under sea-level rise, which will alter nesting conditions for MacGillivray's seaside sparrow. In our individual-based model, MacGillivray's seaside sparrows are predicted to have lower rates of nest success under future sea-level rise conditions (Hunter, 2017, pp. 464-466). While the subspecies exhibits nest-site selection behaviors in response to perceived flooding and predation risks, changes in regular tidal cycles may occur on a temporal scale that is faster than MacGillivray's seaside sparrows can adapt to. Currently, the subspecies' full nesting cycle fits within the approximately 28-day window between two high lunar (spring) tides (Hunter, 2016, p. 133; Hunter, et al., 2016b, p. 136). Future tides that are higher and occur in shorter windows will diminish the ability of MacGillivray's seaside sparrows to nest successfully.

In conclusion, we expect future representation to decline due to the increased risk of genetic isolation from high abundance habitat loss for all three habitat types, and the diminished ability of the subspecies to nest successfully due to anticipated negative changes in predation risk and flood risk.

5.4.3 Future Redundancy

MacGillivray's seaside sparrow currently has four populations spread throughout its range (Figure 4-1). The subspecies has been extirpated from an area in Florida that resulted in a 27% contraction in the historical range, and due to continued mangrove expansion, it is plausible that the MacGillivray's seaside sparrow range may contract further northward in the future. The three South Carolina populations are currently likely stable and have higher resiliency than the Georgia-Florida population, which is experiencing a declining trend in resiliency due to negative population growth rates presumably from low nest survival rates. In the future, redundancy will be reduced because of the loss of high abundance breeding habitat across the range resulting in three of the four MacGillivray's seaside sparrow populations having reduced resiliency (North Inlet-Bulls Island, Headquarters Island, and Georgia-Florida). For the Georgia-Florida population, the current declining trend in resiliency will worsen due to reductions in habitat and nest survival. In addition, the habitat loss could result in habitat fragmentation (further reducing redundancy) leading to isolated pockets of birds along the major river estuaries (particularly Savannah, Altamaha, and Satilla), instead of a single contiguous population as it is currently distributed (E. Hunter, 2017, pers. comm.). In addition, any connectivity between the populations will likely remain low or become further reduced as habitat patches become increasingly fragmented with sea-level rise.

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APPENDIX A

Preliminary Population Genetic Results from Woltmann and Hill "Population genetic assessment of MacGillivray's Seaside Sparrow (*Ammodramus maritimus macgillivraii*) as a potentially threatened taxon."

Preliminary Population Genetic Results from Woltmann and Hill "Population genetic assessment of MacGillivray's Seaside Sparrow (*Ammodramus maritimus macgillivraii*) as a potentially threatened taxon."

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Last updated: 04 January, 2018

NOTE: *Analytical results presented here are as of 15 September 2017 and should be considered preliminary. Although we do not expect these results to change significantly, they could: analyses are still ongoing at this point. We expect to add additional samples in the coming months, and hope to complete mtDNA data collection, both of which may change our interpretation of the situation. The text was last updated 04 January 2018 to clarify that the American Ornithological Society's "Birds of North America" accounts are considered the current taxonomic authority for subspecies in North America.*

Key Results Summarized:

1. To date (15 September, 2017) 337 DNA samples from Seaside Sparrows from Georgia to Connecticut have been genotyped at 15 microsatellite loci.

2. Genetic diversity appears similar to patterns found in a study of Gulf Coast Seaside Sparrows. Genetic differentiation along the Atlantic Coast generally follows an isolation-by-distance pattern, with no evidence of increased differentiation in the putative contact zone (North Carolina – Virginia) between MacGillivray's and Northern Seaside Sparrows.

3. Bayesian clustering in the program STRUCTURE reveals four identifiable genetic clusters: (1) Georgia, (2) Headquarters Island, South Carolina, (3) New York, and (4) all other sampling sites.

4. Headquarters Island, South Carolina consistently emerges as the most significantly differentiated cluster, and a small number of samples from a second site in South Carolina (Yawkey Wildlife Center) are not assigned to the Headquarters Island cluster. Differentiation of Georgia and New York sites from all other sites (besides Headquarters Island) is relatively weak.

5. The patterns recovered here do not match expectations of the currently accepted taxonomic designation and range description of the two subspecies of Seaside Sparrow. We note, however, that the type specimen of MacGillivray's Seaside Sparrow was collected from somewhere near Headquarters Island ("Charleston"), meaning that additional work is needed to clarify its status as a subspecies.

6. Additional data, in the form of additional samples for microsatellite genotyping and additional genetic data from our samples (mtDNA sequences) are needed to better understand the nature and context of the genetic clusters identified here.

A. Goals

In light of the petition to list MacGillivray's Seaside Sparrow (*Ammodramus maritimus macgillivraii*; hereafter *macgillivraii*) for protection under the Endangered Species Act, we sought to collect and analyze population genetic data for populations of Seaside Sparrow from Connecticut to Florida to ask whether population genetic data are congruent with current taxonomic treatment.

B. Background

The Seaside Sparrow was first described by Alexander Wilson (as *Fringilla maritima*) in 1811 from what was later determined to be the coast of New Jersey. In 1834 John James Audubon described the subspecies *macgillivraii* from a site near Charleston, SC, which required that Wilson's *A. maritimus* become the nominate subspecies *A. m. maritimus*. Multiple additional subspecies of Seaside Sparrow were subsequently described in the period from 1888-1940; the taxonomic history of the group can best be described as convoluted. However, once the situation on the Gulf Coast became better resolved, the range of *macgillivraii* was defined as Dare Co., NC, south to either Amelia Island, FL (Griscom and Nichols 1920), or only to Camden Co., GA (AOU 1957). Robbins (1983) provides a concise history of the history and ranges of described subspecies of Seaside Sparrow, and Kale (1983) reiterated that *macgillivraii* is present in Nassau and Duval Cos., FL. Morphologically, criteria for identifying nominate *maritimus* and *macgillivraii* are qualitative (see, e.g., Griscom and Nichols 1920). To our knowledge, quantitative diagnostic criteria are not known. Several writers have commented over time that Audubon's choice of type specimen—a bird in juvenile plumage—was an unfortunate one, as subspecific identification is not considered reliable in this plumage.

Two additional Seaside Sparrow taxa have been recognized on the Atlantic coast, the most famous being the now-extinct Dusky Seaside Sparrow (*A. m. nigrescens*, variously considered a full species or a subspecies), and the Smyrna Seaside Sparrow (*A. m. pelonotus*), a poorly-known group formerly found in Volusia Co., FL, but which is now considered to be extinct (Robbins 1983, Post and Greenlaw 2009). In their Birds of North America (BNA) account on Seaside Sparrows, Post and Greenlaw (2009) consider *A.m. pelonotus* to be synonymous with *A. m. macgillivraii*, and it should be noted that BNA accounts are considered by AOS to be the taxonomic authority on subspecies in North America.

Two other subspecies had previously been proposed along the Atlantic coast. One, "*A. m. shannoni*," was quickly recognized as a fraudulent attempt to subvert *pelonotus* by a collector (H.H. Bailey) known to be dishonest (Hubbard and Banks 1970), and it is thus of no real interest here. The second was a description of "*A. m. waynei*" from Chatham Co., GA, proposed by the well-respected ornithologist H.C. Oberholser (Oberholser 1931). Based, it seems, on the considerations of Griscom (1944), *waynei* was ultimately never accepted by the American Ornithologists' Union, and it was synonymized with *macgillivraii*.

Lastly, it is worth noting that based on mtDNA, the Federally Endangered Cape Sable Seaside Sparrow (*A. m. mirabilis*) has closer affinities to other Atlantic Seaside Sparrows than to those along the Gulf Coast (Nelson et al. 2000). The currently accepted, extant Atlantic clade of Seaside Sparrows thus includes *maritimus* (breeding: VA north to NH and occasionally beyond, non-breeding: south along the entire Atlantic coast to FL), *macgillivraii* (resident: Dare Co., NC south to Duval Co., FL), and *mirabilis* (resident: FL Everglades).

Our best interpretation of the currently accepted distribution of Seaside Sparrow subspecies generally follows that of Robbins (1983; our Fig. 1). Note that many or most individuals of nominate *maritimus* are migratory and winter within the range of *macgillivraii*.

C. Methods and Results

Sampling
We gathered blood samples (obtained via venipuncture and subsequent blood storage in Queen's lysis buffer) from live Seaside Sparrows from all states with a coastal border from Connecticut south to Georgia. Samples were collected by PIs Woltmann and Hill from VA, NC, SC, and GA in May 2017. Samples were provided to us by experienced researchers working in MD, DE, NJ and NY during May-July 2016. We also analyzed several older samples collected by PI Hill from SC in 2000 and with collaborators from CT in the early 2000s. For this report we analyzed 337 samples collected from CT to GA (Fig. 2). We expect to receive a number of samples from northeastern Florida from Carolyn Enloe at Florida Fish and Wildlife Commission, hopefully by October 2017.

For population genetic analyses we initially designated all states as potential "populations," but samples from Dare Co., NC were considered separately from samples taken farther south in NC because we were particularly interested asking questions about the presumed contact zone of *maritimus* and *macgillivraii* near the VA-NC border. We also classified samples from Headquarters Island SC from the two time periods separately in order to ask whether significant genetic drift could cause them to be classified differently in the larger analyses. Nine samples from SC Yawkey Wildlife Center were considered separately from SC Headquarters Island.

Laboratory Methods
DNA extraction and microsatellite genotyping procedures closely followed those described in Woltmann et al. (2014). One locus (Asμ15) from the 2014 study was dropped because it did not amplify consistently in many of the older samples, and we added two other loci (SOSP14, ZoleG03) for a total of 15 loci. All genotype data were tested with MICROCHECKER (van Oosterhout et al. 2004) for evidence of null alleles or potential scoring problems, and with GENEPOP (Raymond and Rousset 2004) to insure that loci were not linked, and that loci conformed to Hardy-Weinberg expectations.

Analyses
Descriptive population genetic metrics are largely similar across all sampled sites (Table 1). The seemingly lower number of alleles (N_a) in the site "SC Yawkey 2000" is likely due to the small number of individuals processed to date from that site (N=9). The ranges of observed (H_o) and expected (H_e) heterozygosity are comparable to those found in Seaside Sparrows along the Gulf coast (Woltmann et al. 2014).

Pairwise differentiation (F_{ST}) between all sampling locations was significant for a majority of comparisons ($p < 0.01$, based on 5000 randomizations of the data; Table 2). A plot of Genetic Distance against Geographic Distance indicates a more or less expected pattern of isolation-by-distance (Fig. 3).

We used the Bayesian clustering algorithms implemented in STRUCTURE (Pritchard et al. 2000, Hubisz et al. 2009) to estimate the most parsimonious clustering of individuals into distinct populations, and to estimate probability of cluster membership of individuals. The results presented here are from a STRUCTURE run where possible K (K = number of clusters) = 1-14, 100,000 replications of burn-in followed by 1,500,000 replications, and 15 iterations at each K. We used an admixture model with correlated allele frequencies, the "loc prior" option, and otherwise default settings.

Mean log likelihoods of each value of K for the full data set are shown in Fig. 4. We consider the best solutions of K to be 2-4. If we consider there to be 2 clusters, the solution is that the samples from SC

HQI are distinct from all other populations sampled (Fig. 5). If we accept 3 clusters, some STRUCTURE runs seem to have a hard time distinguishing whether GA or NY is the next most distinct: about half of the runs chose one of the two as being the next most distinct, and thus when averaged to create the plot, population membership is assigned equally to both groups. When 4 clusters are considered, NY and GA each are distinguished separately (Fig. 5). The larger error bars around the log likelihoods of solutions of $K > 4$ (Fig. 4) make those solutions more difficult to justify and interpret, as STRUCTURE algorithms are known to become unreliable at larger K when the data contain a significant signal of isolation-by-distance (Pritchard et al. 2010).

As recommended by Pritchard et al. (2010), we ran the STRUCTURE models again after removing the most highly differentiated clusters (HQI) to further explore the remaining data. Interpretation of log likelihoods from these runs is not straightforward, as the log likelihoods continue to increase with each K (Fig. 6). However, within this reduced data set, the solution of $K = 3$ is the most biologically justifiable interpretation: the GA and NY sites are distinguished in this solution, just as they are in the full data set (Fig. 7). The solutions $K = 2$ and 4 are difficult to interpret. $K = 2$ suffers from the same issue regarding GA and NY in the analysis of the full data set, and $K = 4$ generally involves identifying a few individuals in the CT samples.

Interpretation
These results indicate a mismatch between the currently defined range of MacGillivray's Seaside Sparrow and the underlying genetic population structure of Seaside Sparrows along the Atlantic coast. We detected no genetic signal of a break or change in allele frequencies between Dare Co., North Carolina and Virginia. Thus, at the very least, we did not find support for defining that area as a range boundary.

We find the main result—that SC HQI is consistently revealed to be the most differentiated population along the entire Atlantic coast—surprising. We note that the result is the same regardless of which SC HQI samples are included in the analyses (2000, 2017, or both; additional data not presented here), and further note that the differentiation between 2000 and 2017 samples from HQI is both small ($F_{ST} = 0.005$) and statistically insignificant ($P = 0.04$; we consider only $P < 0.01$ to be statistically significant here, and more importantly few population geneticists would consider $F_{ST} = 0.005$ biologically significant regardless of possible statistical significance).

From a taxonomic standpoint, we are bound in our broader interpretations to some extent by the fact that Charleston, South Carolina is the type locality of MacGillivray's Seaside Sparrow (HQI is within Charleston city limits). Which means that our finding that birds sampled near that locality are differentiated at some level from all other populations sampled leaves the possibility that MacGillivray's Seaside Sparrow should continue to be recognized, but that the distributional range needs to be examined closer.

From a population genetic standpoint, the interpretation of why SC HQI is more differentiated than any other group along the Atlantic coast comes down to some combination of small population size and little or no immigration from other populations. Fundamentally, small populations are more prone to genetic drift at shorter time scales, and low immigration rates accelerate drift. Clearly, both factors must be at work. Unfortunately it is difficult to imagine *why* genetic drift in this population is outpacing the counteracting effects of immigration, other than the facts that breeding Seaside Sparrows (a) are sparsely distributed in SC (Hill *unpub. data*), (b) have been sparsely distributed in SC for quite some time (Sprunt 1924, 1926, 1927), and (c) must be presumed to exhibit high breeding site fidelity. The same population genetic interpretation applies to GA and NY if we accept 4 clusters in the full data set, but to a lesser degree.

D. Additional Work Needed

We hope to get blood samples from Florida by October 2017, and we should have them processed and incorporated into microsatellite analyses by December 2017.

Additional data needed that were not part of our original research proposal are (1) mtDNA sequences, and (2) additional samples from SC and GA. Microsatellite data are well-suited for picking up relatively fine-scale or potentially "shallow" divergences among populations – this is why they are often the marker of choice for within-species population genetic studies. At present we cannot say how temporally "deep" the divergence of SC HQI is, although it clearly must go back well before the 2000 samples were collected. If the divergence of HQI is relatively shallow (i.e., perhaps tens of generations), we expect mtDNA (ND2) haplotype patterns to be either (a) consistent with patterns from other populations or (b) perhaps less diverse, but still containing common, shared haplotypes from other Atlantic populations. In contrast, if the isolation of HQI goes back hundreds of generations, we may expect to find different haplotypes or different haplotype frequencies, as was found in the southernmost populations of the Texas Seaside Sparrow (*sennetti*; Woltmann et al. 2014). An indication of divergence of HQI at levels detectable by mtDNA analyses would strongly suggest that a serious taxonomic reassessment of the taxon and its distribution would be warranted.

Additional samples from GA and non-HQI sites in SC should help us resolve some uncertainty about how distinct GA is from other populations, and whether other sites in SC cluster with HQI. We know that 9 individuals from Yawkey Wildlife Center (Georgetown Co. SC) do not cluster with HQI in these preliminary analyses. Additionally, samples from birds wintering in GA and SC should allow for us to ask whether wintering birds from further north are truly diagnosable as such.

E. Funding Acknowledgment

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Table 1. Summary statistics of the 12 groups of samples analyzed here. Pop ID refers to sites in Fig 1. Samples from VA north (7-12) are considered putative *maritimus*, and samples from NC south (1-6) are considered putative *macgillivraii*. As of mid-September 2017 we are still awaiting samples from a collaborator in FL. N_A = mean number of alleles (15 loci), H_o and H_e = observed and expected heterozygosity.

Pop ID	Site	N	N_a	H_o	H_e
1	**GA**	25	8.000	0.794	0.792
2	**SC HQI 2000**	31	7.733	0.787	0.802
3	**SC HQI 2017**	30	8.067	0.826	0.802
4	**SC Yawkey 2000**	9	6.867	0.852	0.787
5	**NC Carteret**	39	9.867	0.786	0.824
6	**NC Dare**	27	8.667	0.790	0.808
7	**VA**	35	9.600	0.799	0.812
8	**MD**	34	9.533	0.825	0.826
9	**DE**	23	8.800	0.846	0.812
10	**NJ**	11	7.200	0.800	0.791
11	**NY**	33	8.533	0.806	0.788
12	**CT**	40	9.333	0.810	0.808

Table 2. Pairwise F_{ST} values for all sampling localities (see Pop ID from Table 1 above). Estimates significantly different from zero ($p < 0.01$, tested via 5000 permutations of the data) are underlined and italicized.

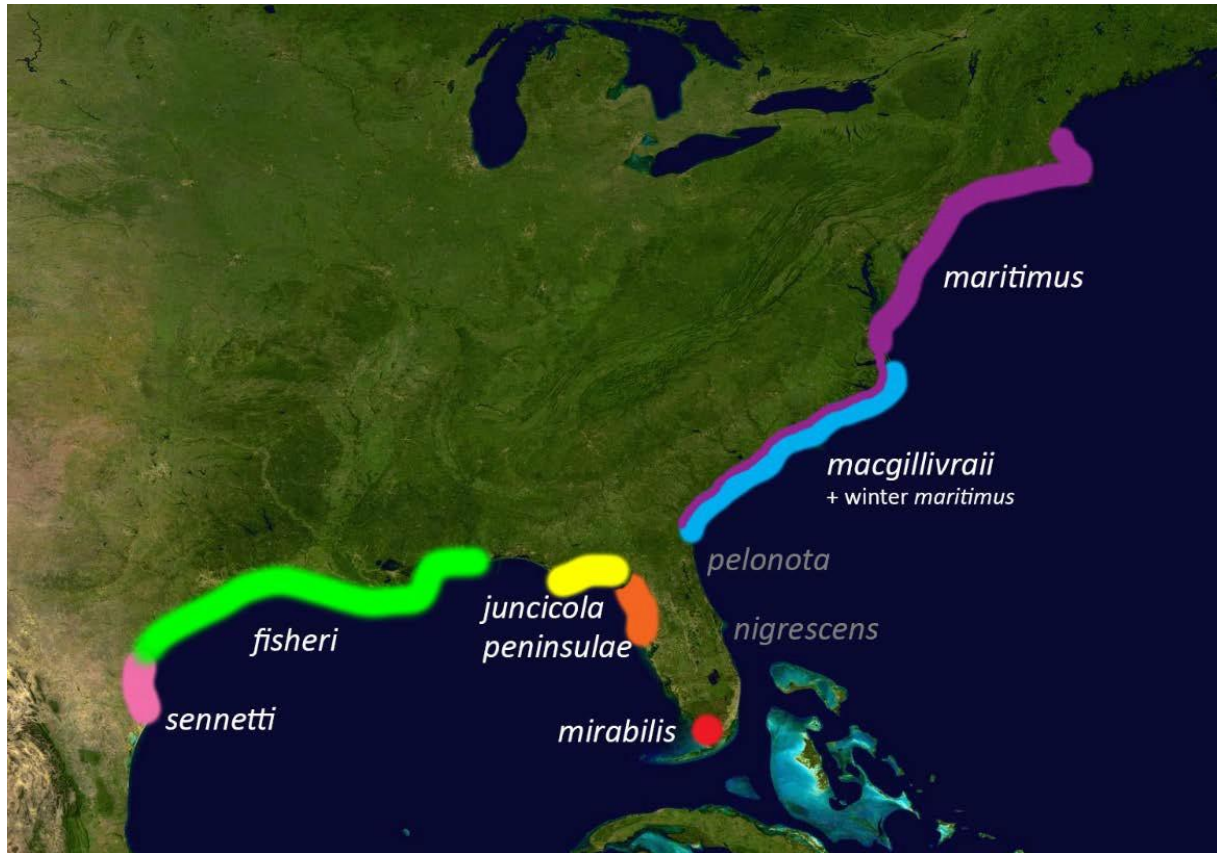|	2	3	4	5	6	7	8	9	10	11	12
1	*0.032*	*0.032*	0.017	*0.025*	*0.022*	*0.020*	*0.014*	*0.026*	0.009	*0.045*	*0.025*
2		0.005	0.015	*0.017*	*0.024*	*0.024*	*0.017*	*0.028*	*0.015*	*0.035*	*0.026*
3			*0.019*	*0.021*	*0.027*	*0.024*	*0.018*	*0.027*	*0.019*	*0.037*	*0.028*
4				0.003	0.000	0.009	0.002	0.014	0.004	*0.023*	*0.020*
5					0.003	0.006	0.003	*0.009*	0.002	*0.021*	*0.010*
6						0.007	*0.003*	*0.012*	0.006	*0.018*	0.007
7							0.000	0.002	0.002	*0.019*	*0.015*
8								0.003	0.003	*0.018*	*0.011*
9									0.007	*0.027*	*0.013*
10										0.011	0.010
11											*0.026*

Fig. 1. Our interpretation of the distribution of subspecies of Seaside Sparrow. Within nominate *maritimus*, many or most individuals winter from VA south to the Atlantic coast of FL. We are uncertain of the subspecific identity of Seaside Sparrows in the western FL panhandle. Note that *pelonota* and *nigrescens* both had very small ranges, and both are extinct.
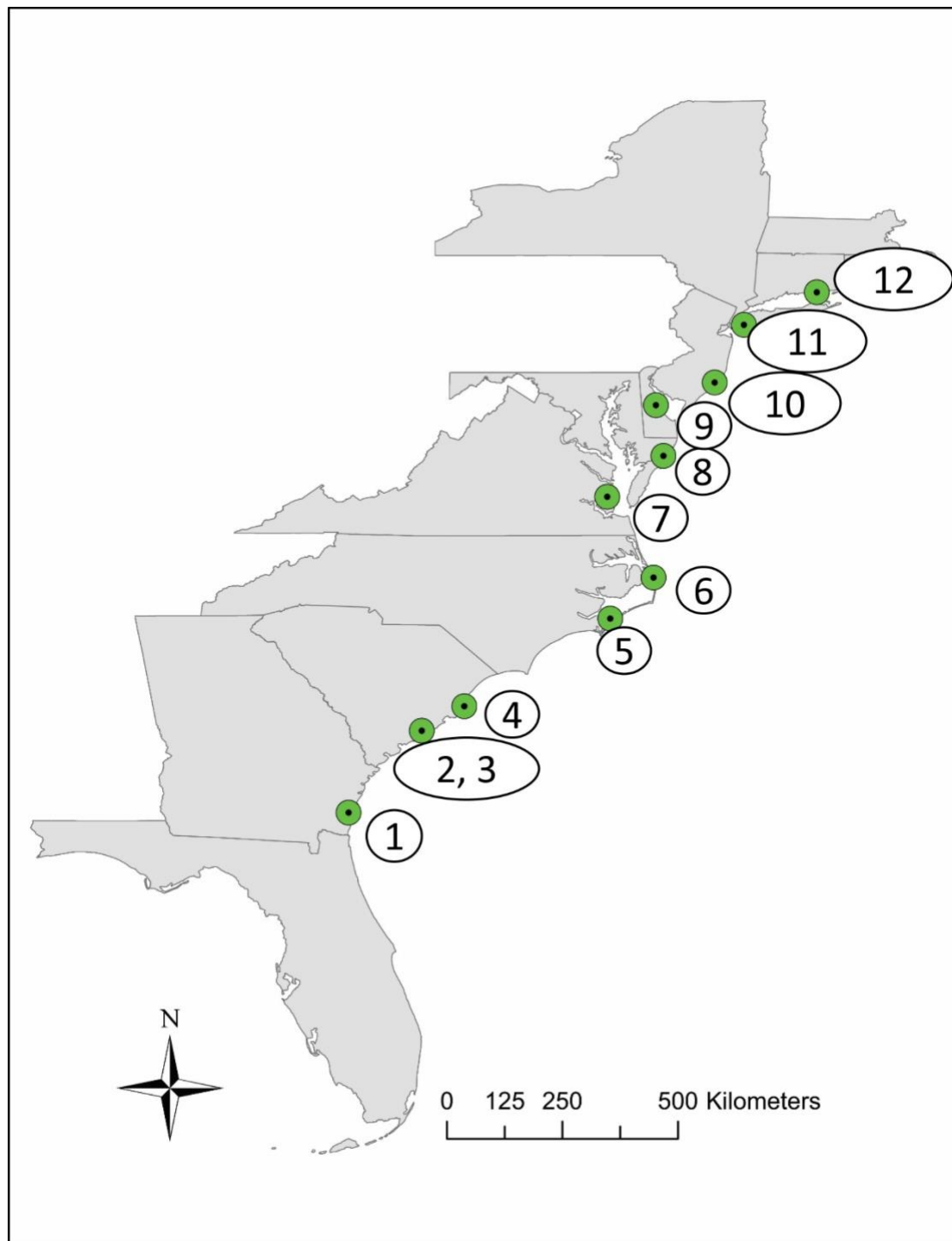
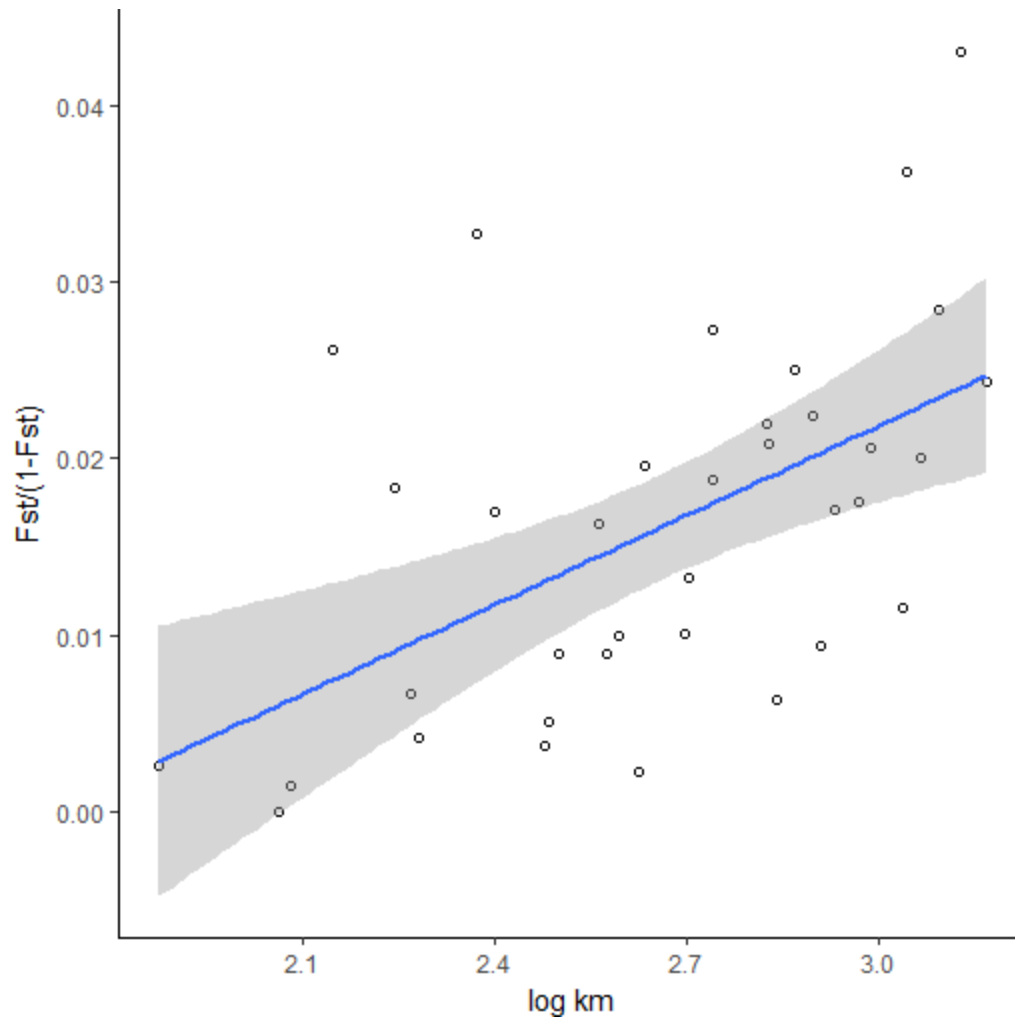Fig 2. Atlantic Seaside Sparrow sampling sites. Numbers refer to Pop ID in Tables 1 and 2.

Fig. 3. The relationship between genetic distance ($F_{ST}/(1- F_{ST})$) and geographic distance (log km), showing a more or less expected pattern of isolation-by-distance (shading = 95% CI; $F_{1,34}$, adj. $r^2 = 0.277$, $p = 0.001$; Mantel test: $p < 0.001$).
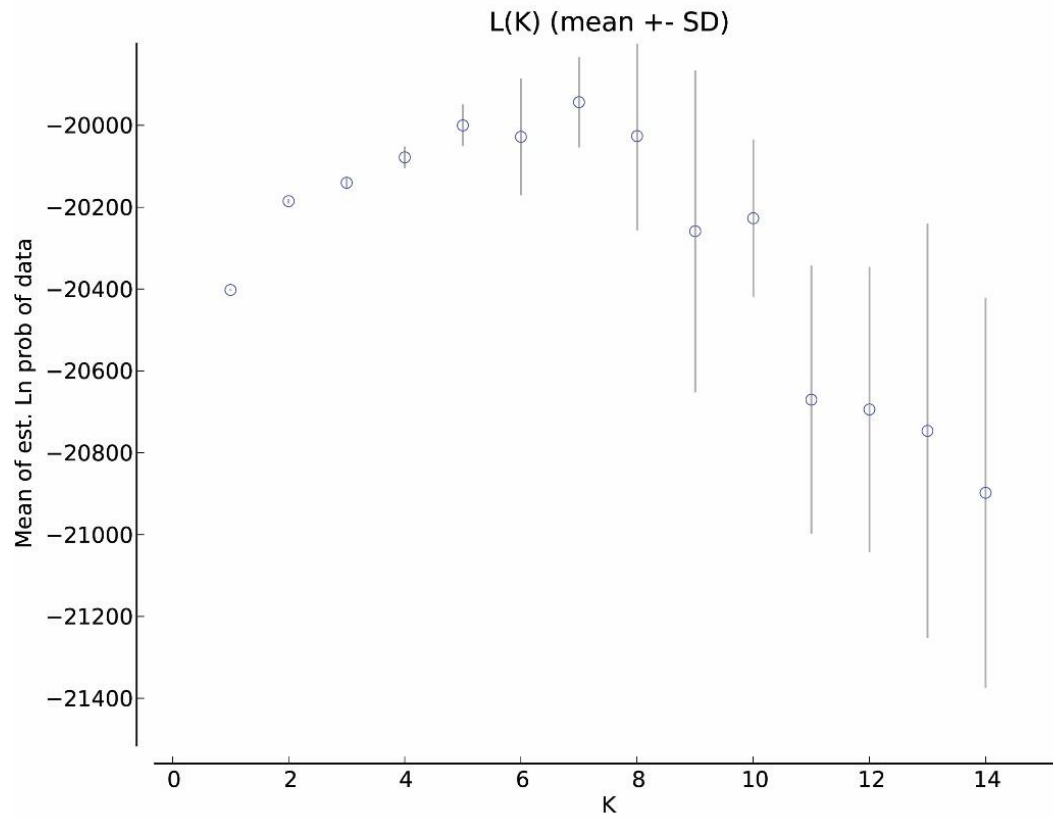
Fig. 4. Mean log likelihoods of STRUCTURE runs to estimate the best number of genetic clusters (K) for all samples processed as of 15 September, 2017. The most conservative estimate is that $K = 2$, but in our opinion the solution $K = 4$ also has merit (see Fig. 5 and text).
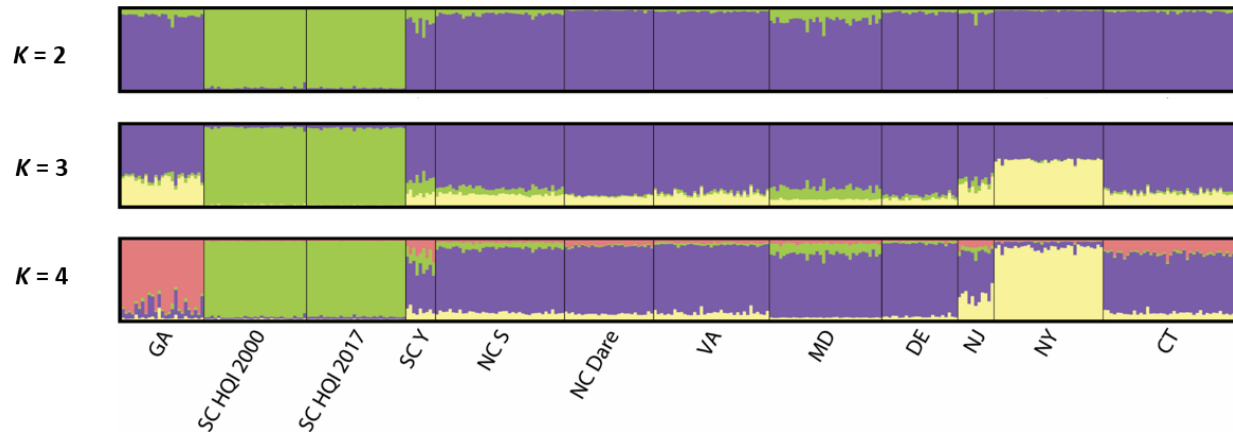
Fig. 5. Results from STRUCTURE showing probabilities of individual assignment to various clusters. In all cases, samples from SC HQI are clearly distinguished from all others. The solution of $K = 3$ is difficult to interpret biologically; we interpret this is an artifact of the algorithms arriving at inconsistent solutions to the question of placing GA or NY next in the hierarchical clustering process: about half of the runs indicated GA as a cluster, and about half indicated NY as a cluster. Please see text for additional interpretation of these three solutions. Note that the putative boundary between Northern and MacGillivray's Seaside Sparrow is between NC Dare and VA.
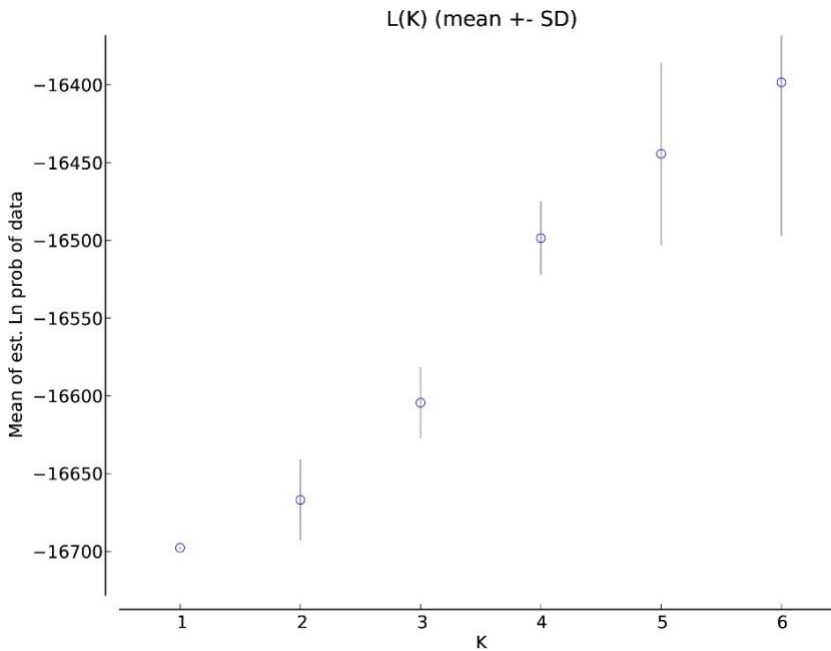
Fig. 6. Mean log likelihoods of STRUCTURE runs to estimate the best number of genetic clusters (*K*) excluding SC HQI for all samples processed as of 15 September, 2017. The most conservative estimate is that *K* = 1, but in our opinion the solution *K* = 3 can be reasonably interpreted from a biological standpoint. *K* = 4 does not have a straightforward biological interpretation (see Fig. 7).
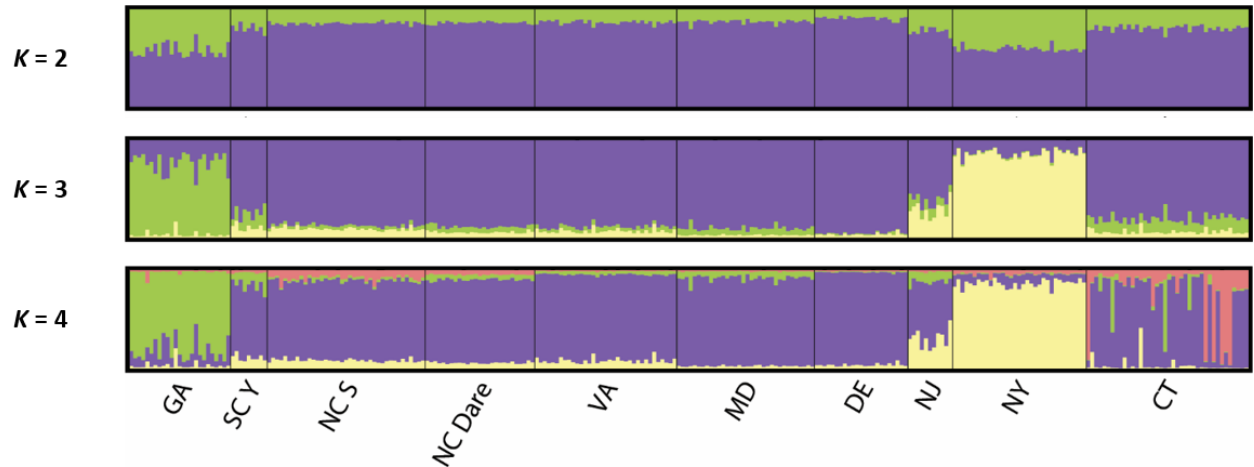
Fig. 7. Results from STRUCTURE showing probabilities of individual assignment to various clusters when SC HQI is omitted from analyses. We consider there to be moderate support in this analysis for distinguishing GA and NY ($K = 3$). Note that the putative boundary between Northern and MacGillivray's Seaside Sparrow is between NC Dare and VA.

Analysis of distribution and abundance of MacGillivray's Seaside Sparrow in South Carolina

Data collected by: Chris Hill, Coastal Carolina University
Analysis conducted by: Elizabeth Hunter, University of Nevada – Reno

November 27, 2017

Methods:
Data used for this analysis were the same as in the final report for the South Carolina State Wildlife Grant SC-T-F13AF01185 (Hill & Laskaris 2015). An occupancy analysis in this report showed no habitat features were correlated with occupancy status. The goal of this re-analysis was to use the same model that found habitat relationships with occupancy and abundance for MacGillivray's Seaside Sparrows (SESP) in Georgia (Hunter et al. 2017). If such relationships could be estimated, state-wide population size could also be estimated (by extrapolating abundances at surveyed sites with similar habitat features). Because of these goals, only habitat features that were available for the entire state were used in analysis; these were: an indicator of salinity, distance to developed areas, distance to forested areas, edge density, proportion of the landscape that is marsh (in a 200m radius around the survey point), and elevation (*see* Hunter et al. 2017 for more detailed descriptions of these habitat variables). Using the original Georgia SESP model on the South Carolina data, no occupancy-habitat relationships were found (in agreement with the original SC analysis), so an abundance-only model (that still accounts for detection probability) was used (this was the same as the model used for Clapper Rails in Hunter et al. 2017). This was a Bayesian model, so estimates are reported as 95% credible intervals.

Results:
Detection probability decreased with increasing noise levels (β = -0.02 – -1.73), but was not affected by date or tidal stage. Higher SESP abundance was associated with greater distances from developed areas (β = 2.05 – 5.83), lower elevations (β = -5.89 – 0.39), and lower salinity marshes (β = 0.36 – 3.04) (Appendix B-1). The relationship between distance from developed areas is the same as that found in Georgia, but the effects of elevation and salinity are the opposite from those in Georgia (SESP in Georgia are found in higher elevation and higher salinity [low] marshes), indicating that SESP in SC are likely using different habitats than those in Georgia. Although these relationships were estimated to be non-overlapping with zero (or nearly so for elevation), there was still substantial uncertainty in them and in the baseline abundance estimate. This uncertainty caused the extrapolated state-wide population size estimate to have such a wide range (from the low thousands to the hundreds of thousands) as to be uninformative. Therefore, these results concur with the analysis in the original South Carolina report that SESP in that state are in small pockets of high densities, but those areas are not predictable based on habitat covariates (or at least those for which there are available data). Other areas outside of those pockets have much lower densities.
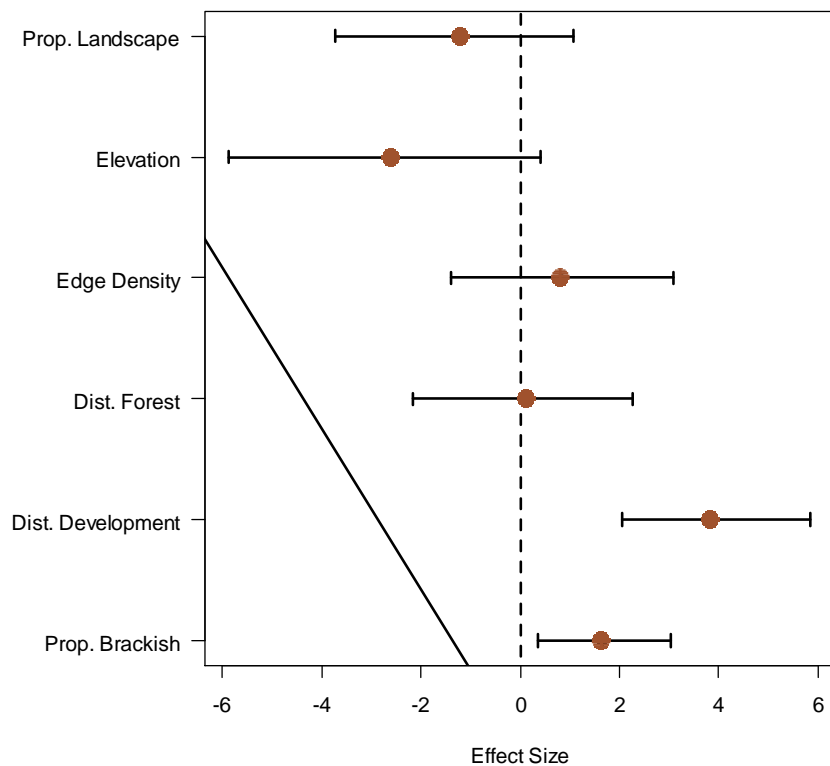
Appendix B-1. Estimated effects of landscape gradients on the distribution of MacGillivray's seaside sparrows in South Carolina. Effect means (points) and 95% credible intervals are shown. Compare to Figure 3 in Hunter et al. (2017).

References:

Hill, C., and Y. Laskaris (2015). Assessing the Status of the MacGillivray's Seaside Sparrows *Ammodramus maritimus macgillivraii* in South Carolina. Final Report to South Carolina Department of Natural Resources for South Carolina State Wildlife Grant SC-T-F13AF01185.

Hunter, E.A., Nibbelink, N.P., and Cooper, R.J. (2017). Divergent forecasts for two salt marsh specialists in response to sea level rise. Animal Conservation 20:20-28.

Modeling population growth rates for three analysis units of MacGillivray's seaside sparrow

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Analysis conducted by: Brett Tornwall and Amy Schwarzer, Florida Fish and Wildlife Conservation Commission

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Methods:
Using nest survival rates reported by individual researchers from populations of MacGillivray's seaside sparrow in South Carolina, Georgia and Florida and adult and juvenile survival rates reported in the literature or derived empirically as described in Table 4-4 of Section 4.3.1 of this report, we ran simulations of potential growth rates (lambda) for each analysis unit. For each simulation we randomly selected from across the possible range of adult survival, juvenile survival and nest survival estimate for that analysis unit and estimated the growth rate for that trial. We performed this calculation 100,000 times in order to develop a distribution of likely growth rate values and then calculated the proportion of lambda values that were less than 1, which would indicate a population decline.

Results:
For South Carolina, 34% of lambda values fell below 1 while 66% of lambda values were ≥1 (Appendix C-1 A), indicating that this unit population is likely stable or rising. For Georgia and Florida, 85% and 79% of trials estimated a lambda below 1 (Appendix C-1 B and C), indicating these analysis units are very likely undergoing population declines under current conditions.
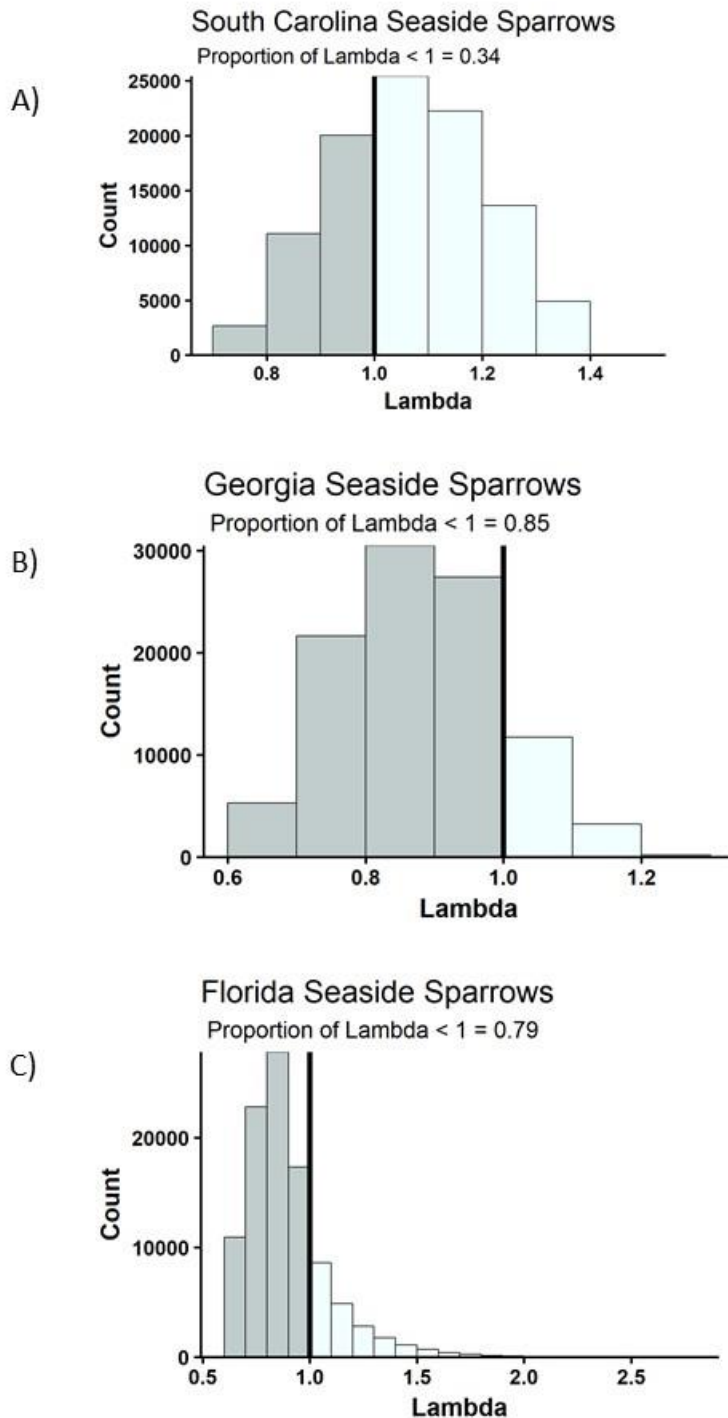
Appendix C-1. Histograms of population growth rate (lambda) simulations for the South Carolina (A), Georgia (B), and Florida (C) analysis units for MacGillivray's seaside sparrow. Gray bars indicate a lambda of <1, light blue indicates a lambda ≥1.